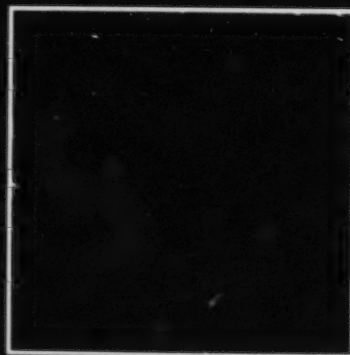
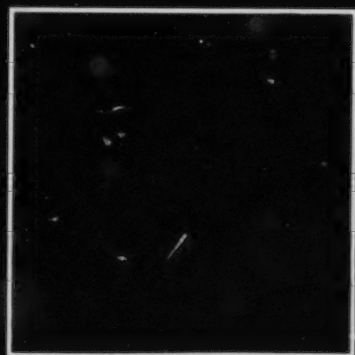


FENITROTHION



Environment
Canada
Conservation and
Protection
Atlantic Region

Environnement
Canada
Conservation et
Protection
Région de l'Atlantique

89-04219/

© Minister of Supply and Services, 1989

Catalogue #En40-370/1989E
ISBN 0-662-16854-2

Published by authority of the federal
Minister of the Environment.

Copies of this report may be obtained from:

Communications Services
Environment Canada
Fifteenth Floor, Queen Square
45 Alderney Drive
Dartmouth, Nova Scotia, B2Y 2N6

(902) 426-7990

environmental effects of

FENITROTHION

use in forestry

impacts on insect pollinators
songbirds & aquatic organisms

Editors* W.R. Ernst, P.A. Pearce and T.L. Pollock

*Pesticides Issues Team
Conservation and Protection
Environment Canada, Atlantic Region

March, 1989



ACKNOWLEDGEMENTS

The Pesticides Issues Team is indebted to many persons, too numerous to mention individually, for assistance in the preparation of this review. Conservation and Protection colleagues at Headquarters provided valuable critical advice on an earlier draft. Technical comment was provided by Forestry Canada staff at Headquarters, the Forest Pest Management Institute, the Maritimes Region, and the Newfoundland and Labrador Region, as well as by provincial officials of the New Brunswick Department of Municipal Affairs and Environment and the Department of Natural Resources and Energy, and the Newfoundland Department of Forestry and Department of Environment and Lands. Sumitomo Canada Limited provided valuable information.

Moirá O'Leary is particularly thanked for her patience in typing the several drafts of the review.



GENERAL TABLE OF CONTENTS

CHAPTER ONE / CHAPITRE UN OVERVIEW / APERÇU

1.0 Preamble / Préambule	4
1.1 Synopsis / Résumé	5
1.2 Recommendations / Recommandations	11

CHAPTER TWO FENITROTHION AND INSECT POLLINATORS

2.0 Summary	16
2.1 Introduction	16
2.2 Effects of Fenitrothion on Pollinators	18
2.3 Effects of Fenitrothion at the Population Level - Changes in Pollinator Abundance	23
2.4 Effects of Fenitrothion on Plant Fecundity	25
2.5 Economic Impact	30
2.6 Legal Issues	31
2.7 Critique	32
2.8 Information Deficiencies	34
2.9 Literature Cited	35

CHAPTER THREE FENITROTHION EFFECTS ON FOREST SONGBIRDS: A CRITICAL NEW LOOK

3.0 Summary	47
3.1 Introduction	49
3.2 Aerial Application of Fenitrothion in Canada: A Historical Perspective	49
3.3 Toxicology	54
3.4 Environmental Residues and Fate	61
3.5 Spray Deposition in the Environment	63
3.6 Avian Impact Assessment Techniques	64
3.7 Routes of Exposure	69
3.8 Data Specific to Fenitrothion Use in Forestry	71
3.9 Population Effects	93
3.10 Conclusions	95
3.11 Acknowledgements	95
3.12 Literature Cited	95
3.13 Appendices	107

CHAPTER FOUR FENITROTHION EFFECTS ON AQUATIC ORGANISMS

4.0 Summary	113
4.1 Introduction	114
4.2 Chemical Analysis	116
4.3 Fate of Fenitrothion in Aquatic Systems	117
4.4 Exposure in the Aquatic Environment	119
4.5 Aquatic Residues Detected by Monitoring Programs	119
4.6 Toxicity of Fenitrothion to Aquatic Invertebrates	120
4.7 Effects on Invertebrate Drift	120
4.8 Effects on Benthic Populations	126
4.9 Probability of Measurable Disturbance	138
4.10 Laboratory Studies with Fish	139
4.11 Field Studies with Fish	148
4.12 Information Deficiencies	154
4.13 Literature Cited	155

CHAPTER ONE

OVERVIEW

W. R. Ernst¹, P. A. Pearce², and T. L. Pollock³

- 1 Environmental Protection
Conservation and Protection
Environment Canada
Dartmouth, Nova Scotia
- 2 Canadian Wildlife Service
Conservation and Protection
Environment Canada
Fredericton, New Brunswick
- 3 Inland Waters Directorate
Conservation and Protection
Environment Canada
Moncton, New Brunswick

CHAPITRE UN

APERÇU

W.R. Ernst¹, P.A. Pearce² et T.L. Pollock³

- 1 Protection de l'environnement
Conservation et Protection
Environnement Canada
Dartmouth (Nouvelle-Écosse)
- 2 Service canadien de la faune
Conservation et Protection
Environnement Canada
Fredericton (Nouveau-Brunswick)
- 3 Direction générale des eaux intérieures
Conservation et Protection
Environnement Canada
Moncton (Nouveau-Brunswick)

TABLE OF CONTENTS

1.0 PREAMBLE	4
1.1 SYNOPSIS	5
1.1.1 Insect Pollinators	7
1.1.2 Songbirds	8
1.1.3 Aquatic Organisms	9
1.2 RECOMMENDATIONS	11

TABLE DES MATIÈRES

1.0 PRÉAMBULE	4
1.1 RÉSUMÉ	5
1.1.1 Insectes pollinisateurs	7
1.1.2 Oiseaux chanteurs	8
1.1.3 Organismes aquatiques	9
1.2 RECOMMANDATIONS	11

1.0 PREAMBLE

Fenitrothion (O,O-dimethyl O-4-nitro-*m*-tolyl phosphorothioate) has been used in forest protection in the Atlantic Region since its first experimental application in 1965. Over that period, environmental impact evaluators from both federal and provincial agencies, as well as academia, monitored and conducted research on operational and experimental spraying of that insecticide. Fenitrothion has been the subject of several reviews by the National Research Council of Canada, the last comprehensive one in 1977. There is now a critical mass of data and a gathering perception that fenitrothion in forest spray use patterns causes impacts that should be questioned, particularly in light of society's increased concern for environmental health.

Recent recommendations of the New Brunswick Committee for Environmental Monitoring of Forest (Insect) Control Operations and advisory correspondence from the federal interdepartmental Atlantic Region Pesticides Advisory Committee reflect concerns about the environmental effects of current fenitrothion use in forestry. Those professional concerns and perceived public unease about the environmental "safety" of that insecticide prompted a decision by Conservation and Protection, Environment Canada, Atlantic Region to sponsor a review of the environmental effects of fenitrothion spraying. The Policy and Planning Committee of Conservation and Protection, Atlantic Region assigned the task of conducting that review to the regional Pesticides Issues Team. The Team focused on insect pollinators and pollination, forest songbirds, and aquatic organisms because of a judgement, shared by a wider body of professional opinion, that most well documented negative effects of fenitrothion spraying are in those environmental compartments. The purposes of the review were: to identify more precisely the magnitude of spray effects in those compartments; to judge whether the identified impacts are ecologically disruptive; to define knowledge gaps; and to determine whether natural resources, which Conservation and Protection has a mandate to protect, are at risk.

1.0 PRÉAMBULE

Depuis les premiers épandages expérimentaux réalisés en 1965, le fénitrothion (thiophosphate de O-O-diméthyle et de O-nitro-4 *m*-tolyle) a été utilisé pour protéger la forêt dans la région de l'Atlantique. Pendant cette période, des représentants tant des organismes fédéraux ou provinciaux que du monde universitaire ont évalué ses répercussions sur l'environnement en effectuant des contrôles et en réalisant des recherches sur la pulvérisation opérationnelle et expérimentale de cet insecticide. Le Conseil national de recherches du Canada a fait plusieurs revues portant sur le fénitrothion; la dernière qui soit complète remonte à 1977. Il existe actuellement une masse critique de données et une impression qui se précise indiquant que la pulvérisation de fénitrothion sur les forêts devrait être remise en question à cause de ses répercussions, en particulier dans l'optique de l'inquiétude croissante qu'a la société pour l'hygiène du milieu.

Les recommandations récentes formulées par le comité du Nouveau-Brunswick chargé du contrôle environnemental des opérations de lutte contre les insectes forestiers ainsi que les avis écrits reçus des représentants de la région de l'Atlantique des différents ministères fédéraux participant au Comité sur les pesticides sont un reflet des craintes croissantes relatives aux effets environnementaux de l'utilisation actuelle du fénitrothion en foresterie. Ces craintes d'ordre professionnel et la perception d'un malaise chez le grand public à propos de la "sécurité" environnementale de cet insecticide ont incité Conservation et Protection, Environnement Canada (région de l'Atlantique) à prendre la décision de parrainer une revue des effets environnementaux des arrosages au fénitrothion. Le comité de Conservation et Protection (région de l'Atlantique) responsable des politiques et de la planification a confié la tâche d'effectuer la revue à l'équipe régionale chargée de la question des pesticides. Cette équipe a concentré ses efforts sur les insectes pollinisateurs et la pollinisation, les oiseaux chanteurs des forêts et les organismes aquatiques; en effet, selon un nombre de plus en plus grand de professionnels, les effets négatifs les mieux connus des arrosages au fénitrothion se font sentir dans ces compartiments de l'environnement. La revue avait pour objet: d'identifier plus précisément l'importance des répercussions ressenties dans ces compartiments; de juger si les répercussions identifiées sont vraiment néfastes pour le milieu; de définir les lacunes dans les connaissances; enfin, de déterminer si les ressources naturelles que Conservation et Protection a le mandat de protéger sont mises en danger.

Separate technical reviews were undertaken, in part under contract, of the effects of fenitrothion on pollinators and pollination, forest songbirds, and aquatic fauna. The reviews were essentially distillations of available literature, with the exception that the songbird review also incorporated previously unpublished data and a re-analysis of earlier published information. Sources accessed included papers published in the primary literature, as well as published and unpublished government reports. The literature examined was extensive but not exhaustive. A selection of only the most relevant material was made and there was no attempt to access proprietary information so as not to preclude release of the final document to the public.

Based on the conclusions of the technical reviews, the members of the Pesticides Issues Team proposed a departmental stance regarding the current use pattern of fenitrothion and developed technical and regulatory recommendations pertaining thereto. The Pesticides Issues Team agreed that a comparison of the environmental effects of fenitrothion with those of other insecticides currently registered for the same purposes, and an assessment of the environmental consequences of not spraying, fell outside the terms of reference for the review but recognized the relevance of such exercises to the development of pest management strategies.

1.1 SYNOPSIS

Because of the considerable continuing interest by environmental investigators in forest insect spray programs, much information has been collected on the environmental effects of fenitrothion. In fact, the environmental data base for fenitrothion developed during the past 20 years of operational use is probably greater than that for any other insecticide in Canadian commerce. One would expect, therefore, that a determination of the environmental acceptability of fenitrothion use would be straightforward. However, there are many difficulties in interpreting complex environmental information. A few of the factors which influence such interpretation are discussed below.

D'autres revues techniques ont été entreprises (en partie sous contrat) sur les effets du fénitrothion sur les pollinisateurs et la pollinisation, les oiseaux chanteurs forestiers et la faune aquatique. Ces revues étaient essentiellement une répétition de la littérature disponible, à l'exception de la revue portant sur les oiseaux chanteurs qui renfermait des données non publiées jusqu'à ainsi qu'une nouvelle analyse des renseignements publiés auparavant. Les sources consultées comprenaient des articles venant des principales publications ainsi que des rapports gouvernementaux, publiés ou non. La liste des publications examinées était considérable, mais la bibliographie n'était pas exhaustive. Seul un choix des matières les plus pertinentes avait été fait, sans aucune tentative d'aller chercher des renseignements exclusifs, de façon à ne pas empêcher finalement la publication du document.

En se basant sur les conclusions des revues techniques, les membres de l'équipe chargée de la question des pesticides ont proposé au ministère une position officielle concernant l'utilisation actuelle du fénitrothion; ils ont aussi mis au point des recommandations techniques et une réglementation à ce sujet. L'équipe des pesticides a convenu qu'une comparaison des effets environnementaux du fénitrothion avec ceux des autres insecticides homologués actuellement aux mêmes fins ainsi qu'une évaluation des conséquences environnementales qu'aurait l'interruption des arrosages ne faisaient pas partie du mandat de la revue; ils ont cependant reconnu la pertinence de ces actions pour ce qui est de l'élaboration de stratégies de gestion des pesticides.

1.1 RÉSUMÉ

À cause de l'intérêt considérable que les chercheurs en écologie continuent de porter aux programmes d'arrosages contre les insectes forestiers, beaucoup de renseignements ont été recueillis sur les effets environnementaux du fénitrothion. En fait, la base de données environnementales mise au point au cours des 20 dernières années sur l'utilisation opérationnelle du fénitrothion est probablement plus vaste que celle concernant tout autre insecticide vendu au Canada. On s'attendrait donc à ce qu'il soit simple de juger de l'acceptabilité environnementale de ce produit. Toutefois, beaucoup de difficultés surgissent quand il s'agit d'interpréter des renseignements complexes sur l'environnement. Suit une discussion sur quelques-uns des facteurs qui interviennent dans ces interprétations.

Firstly, some of the data on fenitrothion impacts have not been collected using classical scientific methods, with study designs that incorporate controls and which are statistically rigorous. That is not a criticism of past monitoring but rather is a recognition that resource constraints were often limiting, and that operational spray priorities took precedence over the requirements of environmental study protocols. Hence, individual studies have produced less clearly defined results than would be desired and they therefore may be somewhat more subjectively interpreted. Such subjective conclusions, as well as the emphasis assigned to them in formulating an overall assessment, may be flavoured by the reviewers' institutional and personal biases. The bias of the Pesticides Issues Team is one of sensitivity to documented risks to the environment. Assessment of the benefits of fenitrothion use is the responsibility of other agencies.

Secondly, difficulty arises in establishing criteria by which negative effects are judged. It is accepted that virtually any use of a pesticide will have negative effects because no pesticide is specific to the target species. Although ecosystems are resilient, some impacts cause disruption of critical ecological processes and alter energy flow, thereby threatening ecological integrity. Aside from those considerations, Conservation and Protection has a responsibility, legislated under the authority of the Migratory Birds Convention Act and the Canada Fisheries Act, to protect wildlife and fish resources from harmful influences. In exercising those mandates, the position has been adopted that environmental impacts which result in direct mortality or in a loss of production of wildlife or fish are not acceptable.

Finally, it must be realized that the technical reviews focused on the forestry use pattern of fenitrothion. Fenitrothion use has changed since its introduction, and that has a bearing on the type and magnitude of associated environmental impacts. Unfortunately, not all investigators documented the conditions of use (e.g. dosage rates, formulation characteristics, and application technology) sufficiently to facilitate a clear comparison among different spray regimes. Spray delivery and meteorological effects have been addressed from a spray efficacy point of view but only poorly in the environmental context.

En premier lieu, certaines des données concernant les répercussions du fenitrothion n'ont pas été recueillies grâce à des méthodes scientifiques classiques, au cours d'études statiquement rigoureuses faisant intervenir des témoins. Nous ne faisons point là une critique des contrôles effectués auparavant; nous reconnaissons plutôt que les ressources imposaient souvent des contraintes et que la priorité opérationnelle des arrosages l'emportait sur les nécessités des marches à suivre des études environnementales. Par conséquent, certaines études ont donné des résultats moins bien définis qu'il ne serait souhaité, dont l'interprétation peut donc être quelque peu plus subjective. Cette subjectivité dans les conclusions ainsi que l'importance qui leur est attribuée dans l'évaluation globale peuvent refléter les biais d'ordre personnel ou professionnel introduits par ceux qui ont fait les revues. Dans le cas de l'équipe chargée de la question des pesticides, le biais vient de sa sensibilité aux risques prouvés pour l'environnement. C'est à d'autres organismes que revient la responsabilité d'évaluer les avantages de l'utilisation du fenitrothion.

En deuxième lieu, des difficultés surgissent quand il s'agit d'établir des critères permettant de juger des effets négatifs. Il est reconnu qu'essentiellement toute utilisation d'un pesticide aura des effets négatifs, car aucun pesticide n'est spécifique de l'espèce cible. Bien que les écosystèmes soient résistants, certains effets entraînent l'interruption de processus écologiques critiques, modifiant ainsi le cycle énergétique et menaçant donc l'intégrité de l'écologie. En plus de ces considérations, Conservation et Protection a la responsabilité, en vertu de la Loi sur la Convention concernant les oiseaux migrateurs et de la Loi canadienne sur les pêcheries, de protéger la faune et l'ichtyofaune de tout dommage. La position adoptée dans l'exercice de ces mandats est la suivante: est inacceptable tout ce qui entraîne directement une mortalité ou une perte de production au niveau de la faune ou des poissons.

En dernier lieu, il faut réaliser que les revues techniques se sont concentrées sur l'utilisation du fenitrothion en foresterie. Les utilisations du fenitrothion ont changé depuis son introduction, ce qui influe sur le type et l'amplitude de ses répercussions environnementales. Malheureusement, les chercheurs n'ont pas tous spécifié suffisamment les conditions d'utilisation (p. ex., taux d'épandage, propriétés de la formulation et mode d'épandage) pour permettre une comparaison évidente entre les divers modes d'épandage. La méthode d'arrosage et l'effet des conditions météorologiques ont été étudiés dans la perspective de l'efficacité de l'arrosage, mais en tenant peu compte du contexte environnemental.

All the reviews note that there is little consistent relationship between operational dosage rate and observed biological effects. One of the possible explanations for the variability in observed biological impact is a difference in susceptibility at the population level. Although it is generally acknowledged that spray accountability is not well understood, it is the opinion of the Pesticides Issues Team that the unevenness of localized insecticide deposits accounts for much of the measured variability in biological response.

The most important conclusions from each of the reviews are presented below.

1.1.1 Insect Pollinators

The pollinator review presents some of the strongest evidence against the use of fenitrothion in forests. Paradoxically, the Pesticides Issues Team notes that this area received the least research scrutiny in the last decade of any of the subject areas reviewed.

- A clear cause and effect relationship has linked fenitrothion use with population decreases of honey bees and wild bees. Furthermore, those decreases caused a disruption of a critical ecological process, as demonstrated by the impairment of plant sexual reproductive success, and have implications for seed- and fruit-eating forest animals.
- Losses in seed production and fruit set led to economic consequences in a managed crop situation, a problem resolved by the establishment of no-spray buffer zones around blueberry fields.
- While the duration of pollinator population reductions in the forest is somewhat speculative, long-term ecological damage in commercial blueberry fields was evidenced by the fact that pollinator population recovery took several years, depending on the history of spray applications in the surrounding area.
- Although there are no data for fenitrothion, studies of the effects of other insecticides on pollinators indicate the strong potential for spray-induced behavioural effects which may compound the ecological consequences of population reductions.

Selon toutes les revues, il n'y a pas grande cohérence dans la relation entre les taux d'épandage opérationnel et les effets biologiques observés. Une des explications possibles de la variabilité des effets biologiques observés est l'existence d'une différence de sensibilité à l'intérieur de la population étudiée. Bien qu'il soit généralement notoire que la responsabilité des arrosages n'est pas bien comprise, l'équipe chargée de la question des pesticides est d'avis qu'une bonne partie de la variabilité des réponses biologiques vient d'irrégularités locales dans les dépôts d'insecticides.

Voici les conclusions les plus importantes tirées des diverses revues.

1.1.1 Insectes pollinisateurs

La revue sur les pollinisateurs présente certains des arguments les plus frappants contre l'utilisation du fénitrothion dans les forêts. Selon l'équipe chargée de la question des pesticides, c'est, de tous les domaines ayant fait l'objet d'une revue, celui qui, paradoxalement, a été étudié le moins à fond au cours de la dernière décennie.

- Une relation évidente de causalité a été établie entre l'utilisation du fénitrothion et la diminution des populations d'abeilles domestiques et d'abeilles sauvages. Qui plus est, ces diminutions ont entraîné l'interruption d'un processus écologique critique, mis en évidence par la diminution de la capacité de reproduction sexuelle des plantes, ayant des répercussions sur les animaux forestiers granivores et fructivores.
- La baisse de production de graines et une moins bonne fructification ont eu des conséquences économiques dans le cas des cueillettes gérées; ce problème a été résolu par l'établissement de zones tampons non arrosées autour des champs de bleuets.
- On ne sait pas très bien la durée des baisses de population des pollinisateurs dans les forêts. Toutefois, dans les cultures commerciales de bleuets, des dommages écologiques à long terme ont été mis en évidence, car le rétablissement des populations de pollinisateurs pouvait prendre plusieurs années selon les épandages qui avaient eu lieu dans les environs.
- Des études réalisées chez les pollinisateurs avec d'autres insecticides que le fénitrothion montrent qu'il est fort possible que les arrosages induisent des effets comportementaux ayant pour conséquence écologique des réductions de population.

1.1.2 Songbirds

Although an understanding of the effects of fenitrothion is incomplete, the evidence indicates that it poses a considerable risk to protected migratory songbirds and casts doubt on the advisability of broadscale spraying of that insecticide in forestry.

- Sporadic observed mortality of the most vulnerable and sensitive songbirds has been associated with fenitrothion spraying of forests, even at the lowest operational dosage rates employed.
- There is evidence that a 50% reduction in brain cholinesterase activity of birds sampled after sprays is a better indication of mortality in the population at large than is the finding of dead or moribund birds.
- In every recent brain cholinesterase monitoring study some birds with at least 50% inhibition were documented. The proportion of sampled birds with that level of inhibition has been as high as 55%. That indicates that mortality in operational spray programs has probably been higher than observations of avian casualties previously indicated.
- Severe reproductive impairment in a typical forest songbird, the White-throated Sparrow, was associated with a mean brain cholinesterase inhibition of 42%.
- Sublethal effects may be as biologically important in contributing to the risk to songbirds from fenitrothion spraying as the direct mortality of birds.
- Most assessment techniques underestimate the impacts of forest spraying on songbirds.
- Although laboratory studies have indicated that captive birds are able to tolerate brain cholinesterase depression of greater than 50%, such investigations are of limited value in predicting impacts in the wild, where birds are subject to complex environmental stresses not reproducible in the laboratory. Such environmental factors are likely to compound insecticide toxicity.

1.1.2 Oiseaux chanteurs

Bien que les effets du fénitrothion ne soient pas connus complètement, les données recueillies montrent que ce produit met considérablement en danger des espèces protégées d'oiseaux chanteurs migrateurs; en outre, elles remettent en question le bien fondé de l'arrosage des forêts en grande échelle au moyen de cet insecticide.

- De temps à autres, on a observé chez les oiseaux chanteurs les plus vulnérables et les plus sensibles, des mortalités associées à la pulvérisation de fénitrothion sur les forêts, même aux taux d'épandage les plus bas utilisés.
- Des données semblent indiquer qu'une réduction de 50 % de l'activité de la cholinestérase dans le cerveau d'oiseaux échantillonnés après des arrosages est une meilleure indication de la mortalité dans l'ensemble de la population que ne l'est le nombre d'oiseaux trouvés morts ou moribonds.
- Au cours de chacun des contrôles récents de la cholinestérase cérébrale, on a signalé une inhibition d'au moins 50 % de cette enzyme chez un certain nombre d'oiseaux. Les proportions des oiseaux échantillonnés chez qui on observait une telle inhibition ont même atteint 55 %. La mortalité au cours des programmes opérationnels d'arrosage a donc probablement été plus élevée que ne le laissait supposer le nombre de décès observés chez les oiseaux.
- Chez un oiseau chanteur typique de nos forêts, le pinson à gorge blanche, des effets négatifs graves sur la reproduction ont été associés à une inhibition moyenne de 42 % de la cholinestérase cérébrale.
- Des effets sublétaux peuvent avoir une importance biologique aussi grande que la mortalité directe pour ce qui est du risque que les arrosages au fénitrothion constituent pour les oiseaux chanteurs.
- La plupart des techniques d'évaluation sous-estiment les répercussions des arrosages forestiers sur les oiseaux chanteurs.
- Des études de laboratoire ont montré que certains oiseaux captifs sont capables de tolérer une diminution de la cholinestérase cérébrale dépassant 50 %; ces recherches n'ont cependant qu'une valeur limitée pour ce qui est de prévoir les répercussions dans la nature où les oiseaux sont soumis à des stress environnementaux complexes qui ne sont pas reproductibles en laboratoire. Il est probable que ces facteurs environnementaux s'ajouteront à la toxicité de l'insecticide.

- Inherent difficulties prohibit the study of the effects of insecticides on forest birds at the population level. Thus, hypotheses that forest songbird populations are affected by fenitrothion spraying cannot be adequately tested.

- Forest structure, topography, meteorological conditions, and factors related to spraying technology influence spray deposit. Those variables, singly or in combination, may cause insecticide deposits greater than average locally, occasioning high levels of exposure to birds.

1.1.3 Aquatic Organisms

The impact of fenitrothion on aquatic systems has been extensively studied, leading to the conclusion that, for the most part, the risks to protected aquatic organisms are tolerable or amenable to mitigation through implementation of buffer zones.

- Aquatic invertebrate responses have been highly variable and not consistently directly related to spray dosage rate. The relationship between dosage rate and biological response is modified by factors such as spray formulation characteristics and application technology, topography, meteorological conditions, forest canopy density, water depth, and stream riffle-to-pool ratios.

- Dramatic benthic invertebrate kills were infrequently observed in lotic systems. Some reductions in benthos were found in 15% of small, unbuffered streams studied during operational spraying. In large lentic systems, invertebrate impacts generally have not been observed.

- In streams, benthic invertebrate depletions were usually short-lived. Most population recoveries occurred within season and are considered to be tolerable.

- In a sensitive environment, bog ponds, up to 50% reductions in total numbers of benthic invertebrates occurred after treatment with fenitrothion at simulated operational dosage rates, and required greater than one year for populations to recover. The ecological implications of such impacts have not been investigated but the potential exists for

- Des difficultés inhérentes empêchent d'étudier les effets que les insecticides ont sur les oiseaux forestiers au niveau de la population. Il n'est donc pas possible de vérifier adéquatement l'hypothèse qui suppose que les populations d'oiseaux chanteurs forestiers sont affectées par le fenitrothion.

- La structure de la forêt, la topographie des lieux, les conditions météorologiques et des facteurs liés aux techniques d'arrosage influent sur la retombée de l'aérosol. Ces variables peuvent, seules ou ensemble, entraîner localement des retombées d'insecticides supérieures à la moyenne, exposant ainsi les oiseaux à des teneurs plus élevées.

1.1.3 Organismes aquatiques

Les répercussions du fenitrothion sur les systèmes aquatiques ont fait l'objet d'études poussées qui ont permis de conclure que, dans la plupart des cas, les risques encourus par les organismes aquatiques protégés sont tolérables ou peuvent être atténués par l'établissement de zones tampons.

- Les invertébrés aquatiques ont eu des réactions très variables qui n'étaient pas toujours liées directement aux taux d'arrosage. La relation entre le taux d'arrosage et la réaction biologique est modifiée par des facteurs tels que les propriétés de la formulation et son mode d'épandage, la topographie, les conditions météorologiques, la densité du couvert forestier, la profondeur de l'eau et les proportions de rapides et de fosses dans le cours d'eau.

- Des hécatombes d'invertébrés benthiques ont été observées de temps en temps dans des systèmes lotiques. Dans 15 % des petits cours d'eau étudiés durant des arrosages opérationnels sans zone tampon, une certaine réduction du benthos fut notée. Dans les grands systèmes lenticques, aucune répercussion n'est généralement observée chez les invertébrés.

- Dans les cours d'eau, les baisses de population d'invertébrés benthiques étaient habituellement de courte durée. La plupart du temps, les populations se rétablissaient à l'intérieur d'une saison, ce qui est considéré tolérable.

- Dans un environnement sensible (des étangs de tourbière oligotrophe), l'épandage de fenitrothion aux taux opérationnels entraînait des réductions de 50 % du nombre total d'invertébrés benthiques, et les populations prenaient plus d'un an à se rétablir. Les répercussions écologiques de ces manifestations n'ont pas été étudiées, mais elles pourraient avoir des effets importants sur la faune vertébrée qui a

significant effects on vertebrate fauna which depend on energy flow through those trophic levels.

- Occasional instances of fish mortality could not be conclusively linked to fenitrothion spraying. A knowledge of concentrations of fenitrothion which are acutely lethal to fish, and maximum measured concentrations of fenitrothion in water after operational spraying, indicate the unlikelihood of spray-induced mortality.

- Fish bioconcentrate fenitrothion. Although residues in fish were high after operational spray programs, they were generally cleared within one day.

- Developmental abnormalities have been observed in fish embryos exposed during operational forest spraying with fenitrothion in Japan at dosage rates much higher than those used in Canada.

- The question of fish population reductions associated with fenitrothion spraying has not been entirely answered. It is axiomatic, however, that they must be less than the aquatic invertebrate reductions, which have been sporadic and small.

besoin de l'énergie accumulée à ces niveaux trophiques.

- Des décès occasionnels de poissons n'ont pu être liés de façon certaine à des arrosages au fénitrothion. D'après les concentrations de fénitrothion qui sont notoirement létales pour les poissons en condition aiguë et les teneurs maximums en fénitrothion mesurées dans les eaux après les arrosages opérationnels, il est peu probable que les mortalités observées aient été causées par ces arrosages.

- Le fénitrothion est bioconcentré par les poissons. De fortes teneurs résiduelles ont été décelées dans les poissons après des programmes d'arrosage opérationnel, mais elles disparaissaient généralement au cours de la première journée.

- Au Japon, des anomalies du développement ont été observées chez des embryons de poissons exposés durant des opérations d'arrosage forestier à des doses de fénitrothion beaucoup plus élevées que celles utilisées au Canada.

- On ne sait pas encore tout à fait quelles sont les réductions de population causées par les arrosages au fénitrothion. Cependant, il va de soi qu'elles devraient être inférieures aux baisses de population d'invertébrés aquatiques, qui étaient elles-mêmes occasionnelles et faibles.

1.2 RECOMMENDATIONS

Many of the studies which have attempted to examine the environmental effects of fenitrothion during the past two decades have suffered from a lack of scientific rigour. That has been partly because of the inadequacies of the tools available to environmental science. However, some of the fault must lie with the dedication of inadequate resources to individual investigations, combined with the logistical difficulties inherent in monitoring spray programs which are conducted according to operational priorities. The individual reviews have nonetheless covered a large number of investigations and the patterns emerging from the cumulative results of those studies allow some judgements to be made on the environmental implications of fenitrothion use. The judgements of the Pesticides Issues Team are encapsulated in the following recommendations. While the Team was satisfied that the overall data base was adequate to draw valid conclusions and was therefore reluctant to recommend yet more studies, a number of important information deficiencies are pointed out.

- Because the accumulated evidence indicates that the present forestry use of fenitrothion causes negative effects on some non-target fauna and on ecological processes of concern to the Pesticides Issues Team, it is judged that large-scale spraying of fenitrothion at registered dosage rates is environmentally undesirable and recommended that an early re-evaluation of the registered fenitrothion use pattern be undertaken.

- Because small lentic systems are sensitive to fenitrothion contamination, it is recommended that increased protection be provided such water bodies by the establishment of appropriate no-spray buffer zones.

- Because fenitrothion is detrimental to pollination of entomophilous plants, a vital ecological process, it is recommended that further insect pollinator studies should be undertaken. Emphasis should be placed on:

1.2 RECOMMANDATIONS

Beaucoup des études réalisées au cours des deux dernières décennies afin de vérifier les répercussions environnementales du fénitrothion manquaient de rigueur scientifique. Cela était dû en partie à l'insuffisance des outils que possédaient alors les écologistes. Toutefois, une partie de la faute doit bien venir de l'application de ressources inadéquates à certaines recherches, en plus des difficultés de logistique inhérentes aux contrôles réalisés dans le cadre de programmes d'arrosage effectués selon des priorités opérationnelles. Chacune des revues a néanmoins ouvert un grand nombre de recherches dont les résultats cumulatifs laissent apparaître des tendances permettant de faire certains jugements sur les répercussions environnementales de l'utilisation du fénitrothion. Les recommandations qui suivent tiennent compte des jugements rendus par l'équipe chargée de la question des pesticides. Toutefois, considérant que l'ensemble de la base de données était suffisante pour tirer des conclusions valables, cette équipe hésitait donc à recommander d'autres études; néanmoins, un certain nombre de lacunes dans les connaissances sont soulignées. La nécessité de certaines données, qui ont été jugées les plus importantes, a été mise à jour par les revues techniques.

- Attendu que la somme des arguments montre que l'utilisation actuelle du fénitrothion en foresterie entraîne des effets négatifs chez certaines espèces fauniques non visées ainsi que sur des processus écologiques que l'équipe chargée de la question des pesticides considère critiques, l'arrosage au fénitrothion en grande échelle aux taux d'épandage homologués est jugé nuisible pour l'environnement, il est donc recommandé qu'on entreprenne rapidement la réévaluation des modes homologués d'utilisation du fénitrothion.

- Attendu que les petits systèmes lenticques sont sensibles à la contamination par le fénitrothion, il est recommandé d'augmenter la protection de ces pièces d'eau en établissant des zones tampons appropriées où aucun épandage n'aura lieu.

- Attendu que le fénitrothion nuit à la pollinisation des plantes entomophiles (un processus écologique vital), il est recommandé que les recherches qui porteront sur les insectes pollinisateurs se fassent principalement dans l'écosystème forestier. Elles devront insister sur les effets sublétaux dont la somme peut avoir sur l'environnement

sublethal effects, which may in aggregate be more environmentally disruptive than lethal ones; the relationship between pollinator abundance and the reproductive success of forest flora, particularly the rarer obligate insect-pollinated plants; and the consequences of reduced production of seed and fruit for forest wildlife dependent on those food resources.

- Because the biological significance of avian cholinesterase inhibition is not completely understood, it is recommended that information gaps be filled with regard to both brain and blood cholinesterase inhibition and in particular the effect on the reproduction of bird indicator species in various fenitrothion-sprayed forest habitats.

- Because fenitrothion spraying for forest protection takes place at a time crucial for the survival of certain wildlife dependent on an aquatic invertebrate food resource, it is recommended that the implications of aquatic invertebrate depletions in small bodies of water for such animals be investigated.

- Because no clear relationships have been established among fenitrothion spray technology, spray dosage rate, localized insecticide deposit, and exposure pathways in non-target fauna, it is recommended that those elements be factored out in controlled environment experiments so that their relative importance to biological impacts may be determined.

des conséquences plus graves que les effets létaux; sur la relation entre l'abondance des pollinisateurs et la reproduction de la flore forestière, en particulier des plantes rares qui doivent nécessairement être pollinisées par des insectes; ainsi que sur les conséquences de la baisse de production de graines et d'une moins bonne fructification pour la faune forestière dont la vie dépend de ces ressources alimentaires.

- Attendu qu'on ne comprend pas complètement l'importance biologique de l'inhibition de la cholinestérase chez les oiseaux, il est recommandé que soient comblés les manques de renseignements concernant l'inhibition de la cholinestérase tant cérébrale que sanguine, en particulier en ce qui a trait à sa relation avec la capacité reproductrice des espèces aviennes choisies comme indicateurs dans divers habitats forestiers arrosés au fénitrothion.

- Attendu que les arrosages au fénitrothion visant à protéger les forêts ont lieu à un moment crucial pour la survie de certaines espèces fauniques qui doivent s'alimenter d'invertébrés aquatiques, il est recommandé que soient étudiées les répercussions qu'ont pour ces animaux les baisses de population d'invertébrés aquatiques qui surviennent dans les petites pièces d'eau.

- Attendu qu'aucune relation évidente n'a été établie entre le mode d'épandage du fénitrothion, le taux d'épandage, les retombées locales d'insecticides et la voie d'exposition des espèces fauniques non visées, il est recommandé que soient élaborées des expériences permettant de contrôler ces éléments dans l'environnement de façon à pouvoir déterminer l'importance relative de leurs répercussions biologiques.

CHAPTER TWO

FENITROTHION AND INSECT POLLINATORS

P.G. Kevan¹ and R. C. Plowright²

¹ Department of Environmental Biology
University of Guelph
Guelph, Ontario

² Department of Zoology
University of Toronto
Toronto, Ontario

TABLE OF CONTENTS

LIST OF TABLES	15
LIST OF FIGURES	15
2.0 SUMMARY	16
2.1 INTRODUCTION	16
2.2 EFFECTS OF FENITROTHION ON POLLINATORS	18
2.2.1 Effects at the Individual Level - Toxicology and Behaviour	18
2.2.1.1 Toxicology - laboratory studies	18
2.2.1.2 Toxicology - field studies	19
2.2.2 Effects on Pollinator Behaviour	22
2.3 EFFECTS OF FENITROTHION AT THE POPULATION LEVEL - CHANGES IN POLLINATOR ABUNDANCE	23
2.3.1 Evidence for Population Reduction	23
2.3.2 Magnitude of Population Reduction	24
2.3.3 Recovery of Pollinator Populations	25
2.4 EFFECTS OF FENITROTHION ON PLANT FECUNDITY	25
2.4.1 Fenitrothion and Pollen	25
2.4.1.1 Pesticide toxicity to pollen: laboratory studies	25
2.4.1.2 Toxicity to pollen and its effect on seed or fruit set	25
2.4.2 Reduction of Fruit Set Caused by Fenitrothion Applications	28
2.4.2.1 Magnitude and scope of effect	28
2.4.2.2 Effect of reduced fecundity on plant population biology	29
2.4.2.3 Effect of reduced fecundity on community function	30
2.5 ECONOMIC IMPACT	30
2.6 LEGAL ISSUES	31
2.7 CRITIQUE	32
2.8 INFORMATION DEFICIENCIES	34
2.8.1 Pollination/Pollinator Ecotoxicology and Biology	34
2.8.2 Socio-Economic and Legal	34
2.8.3 Resource Methodology and Administration	34
2.9 LITERATURE CITED	35

LIST OF TABLES

- 2.1 Percent mortality of caged bees 48 h after treatment with aerially applied fenitrothion at 280g/ha @ 23% ground deposit (data from Sundaram et al. (K.N. Barber, pers. comm.) and Holmes pers. comm.). Numbers in parentheses are sample sizes.
- 2.2 Blueberry crop yields (in tonnes) from New Brunswick and Nova Scotia from 1951 to 1987 (Statistics Canada). The years 1970 - 1977 were affected by fenitrothion spraying in New Brunswick but not in Nova Scotia.

LIST OF FIGURES

- 2.1 Components of pollination as affected by fenitrothion.
- 2.2 Percentage mortality of bees in cages with foliage containing fenitrothion residues (after Johansen and Atkins 1978).
- 2.3 The percentage germination (upper) and rate of tube growth of pollen of *Solanum melongena* treated with fenitrothion (0.05%) (after Regupathy et al. 1973).
- 2.4 The effect of fenitrothion on the percentage of aberrations in pollen mother cells (PMCs) of *Hordeum vulgare* at two stages, seedling (seed spray expt.) and seeds (seed soak expt.) (after Kauer and Grover 1985).
- 2.5 Components of pollination directly affected by fenitrothion, indirectly affected, and presumably affected together with processes known to be directly affected.

2.0 SUMMARY

Fenitrothion is highly toxic to honeybees and all other species of bees upon which it has been tested. In fact, FAO and WHO (1970), although concerned only with agricultural settings, state that "it is toxic to bees and should not be sprayed on flowering crops". Nearly all the literature that we have reviewed, including that from studies in Canadian forests, supports that conclusion. Information on pollinators other than bees and information on non-target insects (many of which are anthophiles) indicate that applications of this broad-spectrum insecticide would likely have ramifications with respect to pollination in any ecosystem in which it was used. Indeed, almost all the literature reviewed here makes reference to reductions in populations of honeybees, wild bees and other pollinators, and non-target insects (which we have not reviewed). The studies in which few effects are noted are either equivocal or problematic.

No information is available on sublethal effects of fenitrothion on honeybees or other pollinating insects, although the adverse effects of other insecticides in sublethal amounts is known.

The review of information on the effects of reductions in pollinator populations on plant sexual reproduction indicates that applications of fenitrothion, and other insecticides, have reduced seed- and fruit-set in various plants (especially blueberries), in agricultural and natural ecosystems. The review concludes that some of the published work is sufficiently careful to demonstrate, beyond reasonable doubt, direct cause and effect links from the primary event of fenitrothion application, through the reduction in pollinator population densities, to the eventual reduction of plant fecundity. Although the effected decline in populations of pollinators is the most important factor in the reduction of the fecundity of some plants, it is noted that fenitrothion also has direct and adverse effects on pollen itself, the ecotoxicological significance of which is probably very minimal in any field situation.

The pollinator population reductions are often large, especially in the weeks immediately following forest-spray applications. Estimates of the time required for populations to recover vary, but some 26 studies have indicated residual effects of depression of pollinator populations even after a period of five years or more. Doubtless, population recovery depends, to some extent, on local conditions, and more especially on the continuing history of spray applications in the surrounding region.

Reduced populations of pollinators have created other ecological effects. Notably, there have been changes in the foraging activity and pollinating efficiency of bumblebees remaining after fenitrothion applications to forested regions. On the one hand, those insects which survive after spraying thrive better as a consequence of lessened competition. On the other, experimental evidence shows that they tend to shift their foraging preferences in a way that may have complex results in terms of their pollination effectiveness for different species of forest plants.

The consequences of reduced fruit- and seed-set may well ramify through the ecosystem to affect populations of frugivores and semiovivores, some of which may also be insectivorous, feeding on pest insects. In the absence of definitive data, such ecological complexities, although logically defensible, must remain only matters for speculation.

The only information available on the economic consequences of the effect of fenitrothion spraying on pollination relates to blueberry production in New Brunswick. Based upon data from Statistics Canada, an estimate presented in the present report suggests a significant average annual decrease of about 665 t in total blueberry production in New Brunswick during the years in which the southern part of the province was extensively sprayed with fenitrothion. However, because of unmeasurable sources of error associated with the statistical procedures, it should be viewed only as a preliminary estimate. Whether fenitrothion has any other socioeconomic effects through its influence on pollination systems in Canada is not known.

2.1 INTRODUCTION

Insect pollination of flowering plants is a keystone process in all biomes of the world (Kevan and Baker 1983). The process inextricably links the vast majority of the biomass of animals (insects) and plants (angiosperms) in a co-evolutionary net that is well over 100 million years old (Crepet 1983; Kevan and Baker 1983). Some of the interactions are highly precise one-on-one relationships, whereas others are very general (Faegri and van der Pijl 1971; Proctor and Yeo 1973; Kevan and Baker 1983; Barth 1985). The components of pollination and related processes are outlined in Figure 2.1. Wind pollination, so common in grasses and some trees, is a derived condition (Whitehead 1969, 1983). Plant reproductive systems which by-pass insect pollination, such as self-pollination and asexual processes (Grant 1984), are also derived conditions.

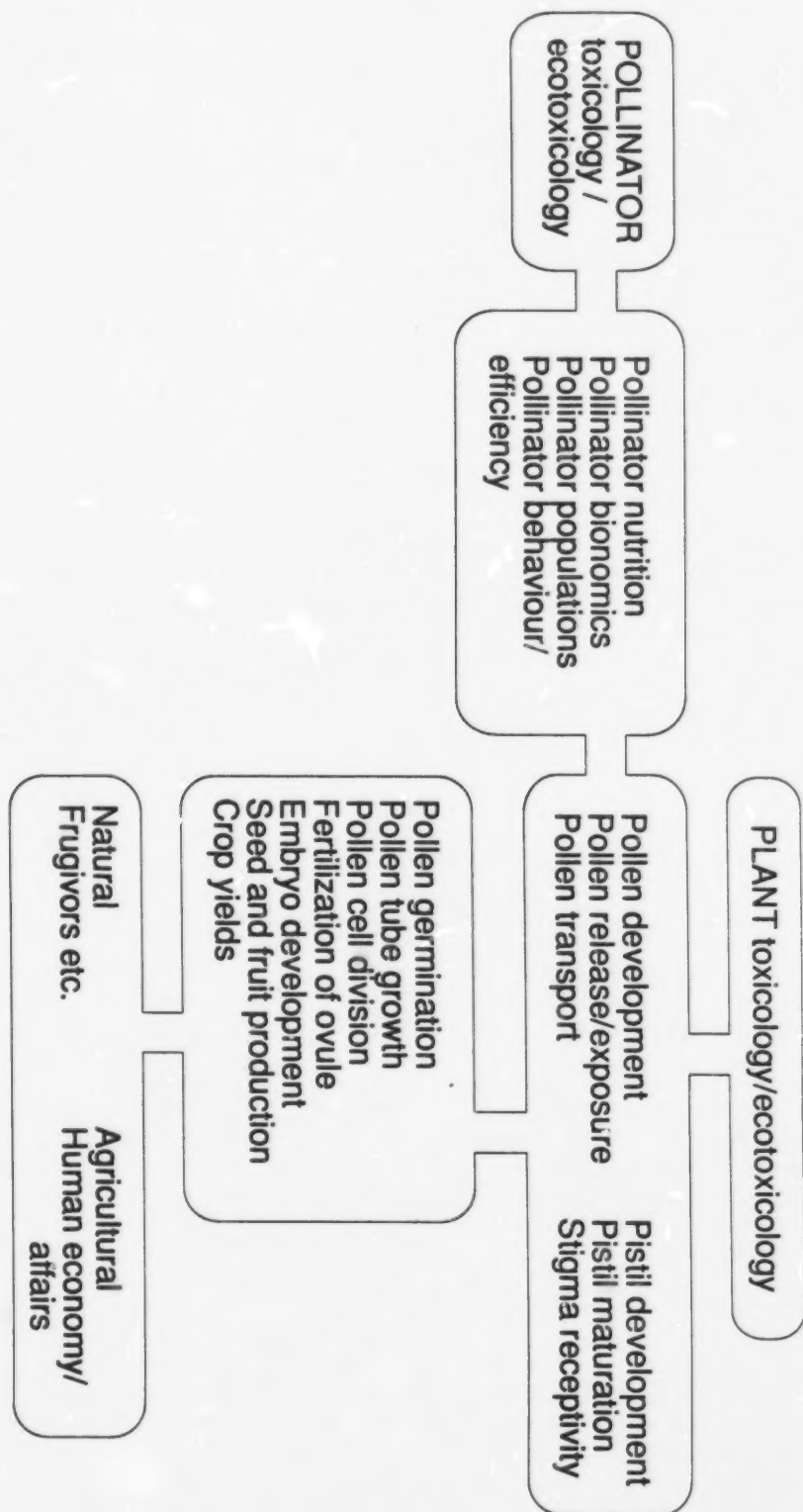


Figure 2.1 Components of pollination as affected by fenitrothion

It is not surprising that the health of the world's biota is strongly dependent on pollination (Kevan 1975a). That is true not only of the natural world but also of the human one in which pollination is crucial to most agricultural productivity (Free 1970; McGregor 1976; Levin 1983). Perturbations of pollination systems in agriculture by insecticides are fairly well documented and, except for some controversy over assessments of their magnitude, are now undisputed (NRCC 1981). Bees, and especially honeybees, are the most valuable insect pollinators to agriculture (Free 1970; McGregor 1976; Levin 1983), and their demise not only evokes concern but also ultimately increases the price of produce.

Fenitrothion is well recognized as highly toxic to bees (Anderson and Atkins 1968; Okada and Hoshiba 1970; Atkins et al. 1970, 1973; Ibrahim et al. 1971; Abdel-Wahab et al. 1973; Gorecki 1973; Hameed et al. 1973; Korpela and Tulisalo 1974; Atkins 1975; Benedek et al. 1976; Plowright et al. 1978a; Stevenson 1978; Capizzi et al. 1979; Plowright and Rodd 1980; Wood 1980a, 1980b; NRCC 1981; Takeuchi et al. 1981; Thakur et al. 1981; Crane and Walker 1983), and its use in Canada and elsewhere led to reductions in pollinator populations in both agricultural (Skrocki 1972; Kevan 1975b, 1975c; Stevenson and Walker 1976; Wood 1977, 1980c, 1980d, 1982a, 1982b; Kevan and LaBerge 1979) and natural (or close to natural) settings (Kevan 1975a, 1975b; Kevan and LaBerge 1979; Plowright 1977; Plowright et al. 1978b; Plowright and Rodd 1980; Bouchard 1981). In fact, in India there are standards with respect to honeybees for formulation of fenitrothion (Indian Standards Institution 1979; Crane and Walker 1983).

Prompted partly by concerns raised by some of the early studies, investigations on fenitrothion were made by the Associate Committee on Scientific Criteria for Environmental Quality (Environmental Secretariat) of the National Research Council of Canada (NRCC 1975, 1977a). The Associate Committee's evaluation of the current status, in 1977, of research on the long-term effects of fenitrothion (NRCC 1977b) contained an unusually strong statement:

The panel was appalled by the generally poor quality of ecological research conducted in conjunction with the fenitrothion spray program. Overall, the program lacks a coherent framework which could serve to link management and research. The majority of the research is pedestrian and is concerned more with data collection than with relevant

experimentation. For these reasons, our understanding of the ecotoxicological impact of fenitrothion has not substantially changed from that arrived at in the previous report NRCC (1975).

The Environmental Secretariat, perhaps influenced by this assessment of the National Research Council of Canada, convened a Panel on Pesticide/Pollinator Interactions. Both Kevan and Plowright, co-authors of this chapter, served on that Panel which published its report in 1981 (NRCC 1981).

For the reasons outlined above, pollination has been recognized as an environmental process that is particularly sensitive to fenitrothion (NRCC 1975, 1977a, 1981). However, despite concerns over the possible long-term effects of its use, it has not been possible to draw unequivocal conclusions. A re-examination of the subject is most timely. Some additional information on the effects of fenitrothion effects on pollination is now available in the form of scientific publications, various reports of government agencies (Canadian, Provincial, and State), theses, and unpublished data. After collecting most of such documents, the information was brought together with previously available material to produce a comprehensive review of the subject.

2.2 EFFECTS OF FENITROTHION ON POLLINATORS

2.2.1 Effects at the Individual Level - Toxicology and Behaviour

2.2.1.1 Toxicology - laboratory studies

The toxicity of fenitrothion has been measured in laboratory studies on honeybees. Okada and Hoshiba (1970) indicate an LD50 of 0.03 µg/bee and an LD90 extrapolated from their data of about 0.1 µg/bee. Atkins (in NRCC 1981, in Johansen and Atkins 1978) used values of 0.176 µg/bee for the LD50 and 0.294 µg/bee for the LD90 although he published higher figures earlier (LD50 of 0.383 µg/bee with probit slope of 4.94) (Atkins et al. 1970, 1973; Atkins 1975; Atkins 1975 based on Anderson et al. 1971). More recently Stevenson (1978) indicated values of 0.018 µg/bee for the contact LD50 and 0.019 µg/bee for the oral LD50.

Helson, Kingsbury, and Barber (K. N. Barber, pers. comm.) found the LD50 for fenitrothion for honeybees

to be 1.213 $\mu\text{g/g}$ of bee (= 0.16 $\mu\text{g/bee}$); 1.814 for *Andrena erythronii* (= 0.10 $\mu\text{g/bee}$); 1.378 for *Megachile rotundata* (= 0.40 $\mu\text{g/bee}$); and 3.978 for *Bombus terricola* workers (= 0.76 $\mu\text{g/bee}$). They measured mortality after 48 h following topical application of the insecticide. The figures in parentheses were converted using some approximations to allow for gross comparison with others' data. The value of the LD50 above, for honeybees, is similar to that used by Atkins (NRCC 1981, see above). Despite that similarity, the reader should be aware that different methods of experimentation produce different results in toxicity studies.

It is agreed that fenitrothion is highly toxic to honeybees and other bees studied. Using the method of Atkins (NRCC 1981) to calculate roughly the dosage of fenitrothion that honeybees would receive at usual rates of application (140-280 g a.i./ha [active ingredient per hectare]) in the field ($[\text{kg a.i./ha}/1.12 = \mu\text{g/bee}]$, and his method (NRCC 1981: Figure 2.4) to estimate the mortality which those rates of application would cause, it is clear that high mortality of honeybees would take place (as documented). We note that the methods used above are for agricultural conditions in California and that in the forest itself, but not in clearings, much fenitrothion would be intercepted before reaching forest herbs and foraging bees. Nevertheless, the hazard is sufficiently great, and poisonings well documented (see below) in forest and near-forest environments in Canada demonstrate that the methods are relevant and cannot be discounted.

There is little information on the toxicity of fenitrothion to other kinds of pollinators. Wood (1980b, 1980e) reported that the LC50 for inhaled doses was 150 $\mu\text{g/m}^3$ for *Apis mellifera* and 190 $\mu\text{g/m}^3$ for *Bombus* workers. Johansen's study (Johansen and Atkins 1978) on *A. mellifera*, *M. rotundata*, and *Nomia melanderi* shows that toxic residues of fenitrothion (albeit applied at dosage rates two and four times the maximum used for single application in forestry) persist on vegetation, to render the environment unsafe to bees for several days. That study, submitted to the Environmental Protection Agency, resulted in a labeling recommendation prohibiting fenitrothion use on alfalfa within 7 d of bloom (Johansen and Atkins 1978). Although those data are not from forest environments, the general effects can be assumed to hold true. Figure 2.2 shows the results of Johansen's study, in which the greater hazard to smaller bees, *Nomia* and *Megachile*, than to honeybees is demonstrated. The results of Helson, Kingsbury and Barber (K. N. Barber, pers. comm.) also follow the same trend. Although that trend is

widely cited (Johansen and Mayer 1976), it must be treated with great care as many exceptions are known (Torchio 1973, and see below).

The physiological measure of risk can be compared among species by effects related to $\mu\text{g/g}$ of body weight. The data of Helson et al. (K.N. Barber pers. comm.) for *Megachile*, *Andrena*, and *Apis*, with relative weights of 1:2:5 show similar LD50s for fenitrothion of 1.378, 1.814, and 1.213, but show that *Bombus* workers, 6.8 times the weight of *Megachile*, had a much higher LD50 of 3.978 $\mu\text{g/g}$. One would expect that smaller amounts of insecticides would be required to kill smaller bees. However, various other parameters increase the level of risk: e.g. leafcutting bees will be more at risk from residues on foliage than will honeybees or bumblebees; bees foraging at upwardly opening flowers with exposed sporophylls and nectar will be more at risk than those foraging at enclosed or downwardly opening flowers with protected sporophylls and nectar (see discussion in Kevan et al 1984); and pollinators foraging in exposed environments (the forest or crop canopy, or edges) will be more at risk than those foraging beneath a blanket of foliage. It is important in such studies as those discussed above and below that the biology of the insect be considered in terms of the insecticide exposure it may experience and that residue concentrations should be expressed in both $\mu\text{g/g}$ and $\mu\text{g/insect}$.

In general, the effects of insecticides on bees, other than European honeybees, are similar (as indicated by Wood's data above for fenitrothion; see also Johansen and Eves 1967; Torchio 1973; Bai and Reddy 1977a); and the values for lethal doses for honeybees can probably be used at least as rough guidelines.

2.2.1.2 Toxicology - field studies

The evidence relating the toxicity of field applications of fenitrothion to mortality of individual pollinators is based on counts of dead individuals, exposure-cage results, and residue analyses. Although mortality is almost always indicated, the results of those field studies are not always easy to interpret. Canadian references are: Sundaram 1974, 1975; Buckner 1974, 1975; Kevan 1975b; Buckner et al. 1975; Buckner and Sarrazin 1975; NRCC 1975, 1977b, 1981; Wood and Stewart 1976; Wood 1977, 1979, 1980d; Plowright 1977, 1980; Buckner and McLeod 1977; Plowright et al. 1978a; Kevan and LaBerge 1979; Plowright and Rodd 1980; Sundaram et al. unpublished.

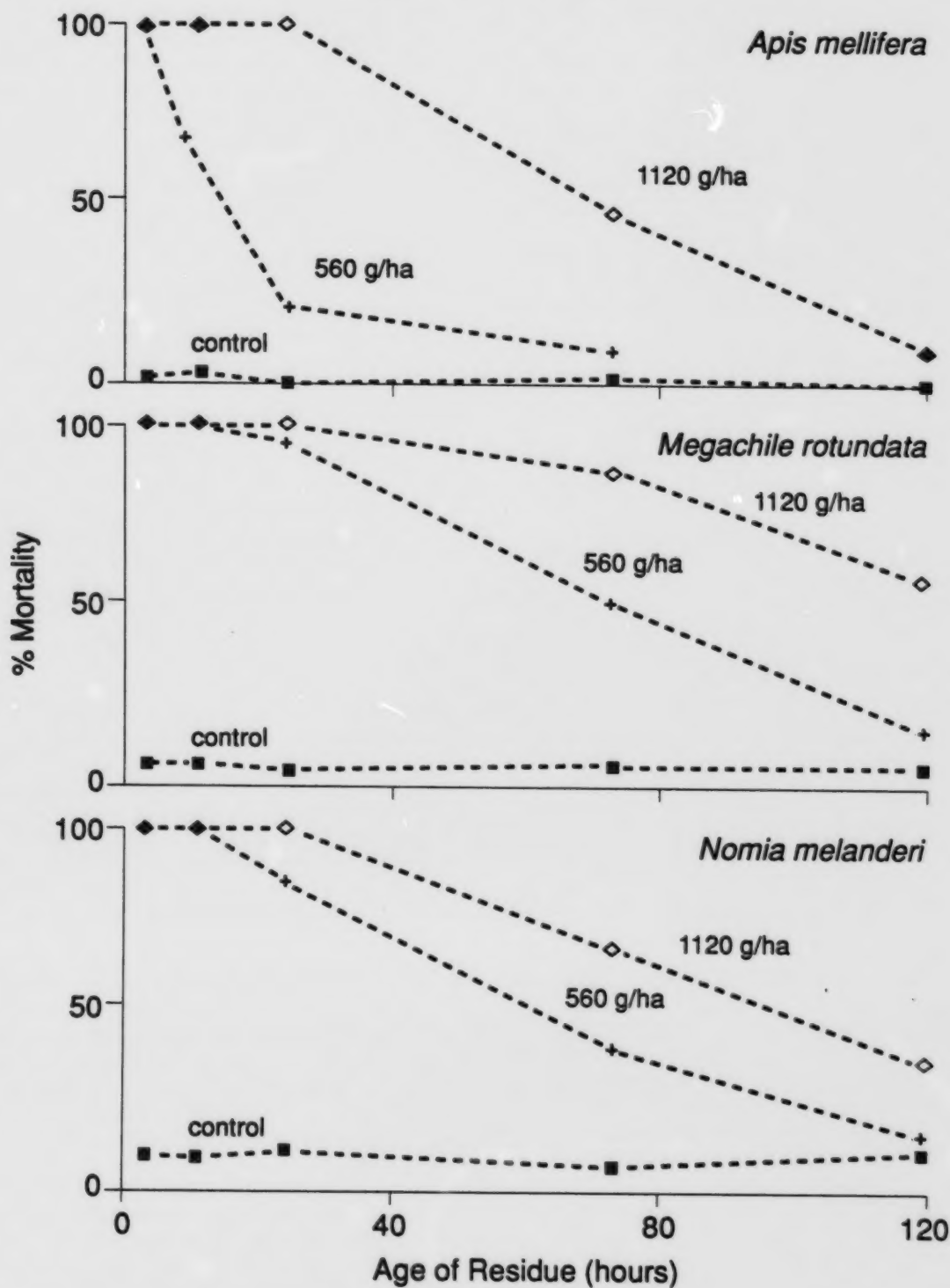


Figure 2.2 Percentage mortality of bees in cages with foliage containing fenitrothion residues (after Johansen and Atkins 1978).

Buckner (1974, 1975), McLeod and Mortensen (1975), and Buckner and McLeod (1977) conclude that there is a minimal effect of fenitrothion on honeybee colonies experimentally placed in areas sprayed with the chemical. However, they recognized mortality in foraging bees. For example, the results tabulated in Buckner and McLeod (1977) showed a decline in the foraging activity of honeybees in fenitrothion-treated plots compared with control plots. They also recorded much higher numbers of dead bees collected at the hive (ca. 120 for each of 3 d after treatment) in the treated area (vs. 12 for the control area). Buckner (1974) made similar counts of dead bees. The results of those experiments are difficult to interpret. The reassuring conclusions of the authors notwithstanding, the results may, in fact, indicate serious losses to the foraging component of the hive.

If one assumes that an individual bee makes between 10 and 25 foraging trips per day, the loss of 120 bees represents 1200 to 3000 trips. Moreover, most bees killed by organophosphorus insecticides probably die in the field before returning to the hive (Atkins 1975), and by the experimental technique of Buckner and his co-workers, would not be detected. Thus, the loss to the colony may be substantial, even though the colony has the capacity to recover quickly by recruitment of foragers and even though, at the time when the studies were made, there was a high rate of oviposition and brood production.

Rigorous field trials of the sort needed have not been undertaken and should be far more complex in design and scope than are exemplified by the studies of Buckner and his colleagues. Studies under the guidelines of the International Commission for Bee Botany (1980), and incorporating the methods of Bacilek (1982) have not been made. The methodology of Bacilek's study is a fine example of experimental rigour: honeybees were trained to use a specific flight path over which the insecticide was sprayed after training was complete. Using that procedure, Bacilek recorded a 41.1% kill of flying bees exposed to fenitrothion spray. It is unlikely that a beekeeper would consider such losses inconsequential.

To offset one of the problems mentioned above - that counts of dead individuals at the hive cannot provide an estimate of mortality of foraging individuals which die while still in the field - Plowright and co-workers developed an experimental "exposure cage" technique for assessing pollinator mortality under field conditions. They exposed captive insects in screened cages in areas sprayed with fenitrothion, while simultaneously setting out other cages containing control individuals in nearby unsprayed

areas. For all insect taxa studied, the results (Plowright 1977; Plowright et al. 1978a; Plowright and Rodd 1980), were decisive. Mortality was mostly high (up to 100% in cages with screen tops placed on the ground in areas lacking dense canopy directly under the path of spray planes) and, in cases where the insects survived, the cage in question happened to be at a distance (ca. 100m) from the closest spray pass. Helson et al. (K.N. Barber, pers. comm.) obtained similar results in a more recent and preliminary series of exposure-cage experiments - mortality of caged bees (in cages which allowed flowthrough of the insecticide spray and which were suspended 0.5 m above the ground in an open area) was total or over 80% in all trials except the control (Table 2.1). The methods used by both Plowright and co-workers and Helson et al., by which bees were placed in unnatural circumstances and in contact with contaminated surfaces (the cages), were not identical: the cage types were different, the dosage rates were 210 vs. 280 g/ha, the application equipment was boom-and-nozzle as opposed to Micronair, and the rates of ground deposition were 10% (estimate from Holmes pers. comm.) for the earlier, vs. 23% for the later study. But despite those differences, the results are remarkably similar.

The importance of knowing what pesticides, and in what amounts, are present in bees killed by pesticides is universally recognized. Methods for the analysis of fenitrothion are well worked out (Mikolajczak and Bohosiewicz 1971; Sundaram 1974, 1975; Bai and Reddy 1977b). The most comprehensive accounts of fenitrothion residues in honeybees are by Sundaram (1974, 1975). He used living and dead bees, pollen, honey, and beeswax collected from hives. The maximum concentration of fenitrothion found in honeybees was 2.08 ppm, and the maximum concentration in pollen was 0.175 ppm. Concentrations in honey and wax were either below or close to the limits of detection (max. 0.055 ppm for one wax sample). Although fenitrothion is rapidly degraded, within a few days, in honeybees and is present in only small amounts absolutely, it is clear from its toxicity (see above) that the concentrations reported by Sundaram (1975) are evidence that fenitrothion was the cause of mortality.

Kevan (1975b) reported fenitrothion residues in dead bees taken from hives in blueberry fields exposed to fenitrothion in New Brunswick. In fields presumed directly affected by fenitrothion, residues from bee carcasses ranged from 0.06-0.95 (mean = 0.261, $n = 7$) ppm fenitrothion, whereas from fields distant from fenitrothion applications, residues ranged from 0.00-0.03 (mean = 0.015, $n = 2$) ppm. In one individual of *Andrena* sp., which was moribund when caught,

the residue concentration was 1.05 ppm. It is assumed that fenitrothion caused the deaths in that study. In another agricultural context, Renval (1977) documented fenitrothion residues in killed honeybees in Sweden.

Sundaram et al. (unpublished) recorded fenitrothion residues in bees caged in experimentally treated plots with little or no interceptive canopy in Ontario to range from 1.35–4.99 ppm, or 426–1659 ng/bee for bumblebees; from 0.87–3.47, or 24–113 ng/bee for leafcutting bees; and from 0.32–1.57, or 44–217 ng/bee for honeybees. In all trials, except one, all bees died within 48 h; and the exceptions were the survival of three of 19 bumblebees and one of 20 leafcutting bees in the case when residues were at their lowest, and one of 20 leafcutting bees when residues were at their second lowest (Table 2.1). They also recorded residues in non-caged insects collected in the wild within 9 h of an aerial application of fenitrothion. They showed similar concentrations: bumblebees (1.04–1.48 ppm; 379–465 ng/bee); solitary bees (mostly *A. erythronii*) (0.96 ppm; 54 ng/bee); and butterflies (mostly *Erynnis* sp.) (8.3 ppm; 569 ng/insect).

2.2.2 Effects on Pollinator Behaviour

Two ways in which insecticides can modify pollinator behaviour, direct and indirect, are considered here.

Sublethal doses of some pesticides have adverse effects on insect behaviour because they interfere with neural and muscular coordination and learning (Schricker 1969, 1972, 1974a, 1974b; Schricker and Stephen 1970; Stephen and Schricker 1970; Cox and Wilson 1984, 1987; Taylor et al. 1987). The work of Schricker (1969) on effects of the organophosphorus pesticide parathion, which is more toxic than fenitrothion to honeybees, is especially relevant. He

first described the effect of a sublethal dose of that insecticide on the ability of honeybees to indicate distance by the dance language. Stephen and Schricker (1970) and Schricker (1974b) found that directionality of the dance language was upset. Later, Schricker (1972) noted adverse effects of sublethal doses of that insecticide on both the chronological memory and the sense of time (Schricker 1974a) in those insects. The effect of other insecticides on honeybee behaviour received attention. Cox and Wilson (1984, 1987) discussed permethrin effects on the general behaviour of honeybees. Taylor et al. (1987) noted that the behavioural conditioning of honeybees is impaired by sublethal doses of synthetic pyrethroid insecticides.

It appears that no work with fenitrothion comparable to that of Schricker on parathion has been done. That creates a distinct gap in knowledge because, in the context of the total effect of insecticides on pollination, it is likely that only a part of the story is given by mere pollinator mortality statistics.

Although fenitrothion depresses pollinator populations, making pollination less frequent (Plowright 1977; Plowright et al. 1980; Thaler and Plowright 1980) as a direct consequence of pollinator scarcity, more subtle effects resulting from shifts in pollinator behaviour in response to ecological changes (that are themselves secondary outcomes of that self-same pollinator scarcity) have been documented. Plowright et al. (1980), for example, found that the rate of removal of pollinia from the lady's slipper orchid, *Cypripedium acaule*, was significantly less where fenitrothion had been applied to an area of the forest habitat with a sparse canopy than in similar nearby untreated areas. They attributed that difference primarily to direct mortality of bumblebees caused by the insecticide, which had been documented

Table 2.1 Percent mortality of caged bees 48 h after treatment with aerially applied fenitrothion at 280g/ha @ 23% ground deposit (data from Sundaram et al. (K.N. Barber pers. comm.) and Holmes pers. comm.). Numbers in parentheses are sample sizes.

Taxon	Control	Sites				Total 1-4
		1	2	3	4	
<i>Apis mellifera</i>	15(40)	100(20)	100(20)	100(20)	100(20)	100(80)
<i>Megachile</i> spp.	0(40)	95(20)	100(20)	100(20)	95(20)	98(80)
<i>Bombus</i> and <i>Psithyrus</i>	38(40)	84(19)	100(14)	100(18)	100(14)	95(80)

together with reductions in their populations (Plowright et al. 1978a; Plowright and Rodd 1980). What makes that case more complex is that *C. acaule* does not offer the bees any reward, and so is presumably visited by pollinators only in the course of their incidental explorations of the flowers (Heinrich 1979). If other flowers in the area were replete with rewards, as Plowright and Rodd (1980) showed for the forest lily, *Clintonia borealis*, then *C. acaule* would likely be ignored to a greater extent than could be explained merely by the reduction in bumblebee numbers. The effect would be to exacerbate the degree of impact of fenitrothion spraying on *Cypripedium* pollination.

The example described in the previous paragraph is a special case of a more general phenomenon - shifts in pollinator behaviour brought about by freedom from competition, especially intraspecific competition, may result in substantial non-linearities in the relationship between pollinator population reduction and consequent depression of fecundity in the plants that those pollinators serve. In the context of forest spraying in New Brunswick, Plowright et al. (1978a) found a striking case of such shifts in behaviour. Bumblebees, from two artificially reared colonies, foraging in habitats that had earlier been sprayed with fenitrothion showed much "narrower food-niches" (i.e. they foraged on fewer plant species) than bees from a colony in comparable unsprayed locations. What that means, presumably, is that in contrast to the case of *Cypripedium*, those plant species which offer the richest rewards to their pollinators are likely to be the least affected by fenitrothion application - the impact on their seed production is expected to be disproportionately small.

Further complications result from other secondary consequences of changes in pollinator behaviour. For example, Thomson and Plowright (1980) and Galen and Plowright (1985) demonstrated in two quite unrelated plant species that the amount of nectar present in a flower determines how long a pollinator will remain on it and, hence possibly how much pollen is picked up on the insect's body. In that way a link between insecticide spraying and pollen carryover (a crucial factor in the dynamics of insect pollination — Plowright and Hartling 1981) can be discerned. Once again, significant non-linearities are introduced into the relationship between pollinator population density and seed set.

The reader will note that much of the work reviewed above concerns honeybees. Honeybees may not represent the ideal pollinator for monitoring the effects of pesticide applications on pollination in forestry. It has been suggested that because

honeybees prefer to forage in open areas and their hives are placed in exposed places they are subject to greater risk than pollinators of the true forest. Therefore, measured insecticidal effects on honeybees may overestimate the impact on wild pollinators. To counter that, we point out that honeybees are the best known insects from the viewpoint of the effects of insecticides, and that in general one can extrapolate from honeybees to other pollinating bees with some confidence. Also, native "forest pollinators" are in fact mostly encountered in open areas such as burns, small cuts, blow-downs, riparian stands, rights of way, road and track edges, etc. Furthermore, the impact on solitary bees, and bumblebee colonies containing only the single foundress queen or a small worker force which cannot be replaced quickly (c.f. honeybees), could result in a far more serious loss to the population and threat to its reproductive success than the honeybee studies would suggest. We submit that whatever their deficiencies honeybees remain the most useful bio-monitors for assessing the effects of pesticides on pollinators. Bumblebees and leafcutting bees are also very useful because they can be manipulated. Fossorial solitary bees are difficult to use because of their nesting requirements and other aspects of their biology.

2.3 EFFECTS OF FENITROTHION AT THE POPULATION LEVEL - CHANGES IN POLLINATOR ABUNDANCE

2.3.1 Evidence for Population Reduction

Some differences of opinion arise in this area. For example, the 1981 Annual Report of the Edinburgh School of Agriculture (1982) included an account of the death of bees (*A. mellifera*) caused by fenitrothion applied to raspberries in bloom. Yet in another example, Romankow et al. (1979) indicated that pollinator populations (of honeybees and wild bees) were not depressed in fields of blooming alfalfa treated with fenitrothion. Neither report provides adequate data to allow for even a tentative explanation of the difference.

In Canada also, in some early studies, either negative or equivocal findings were reported. For example, Varty and Carter (1974) concluded, on the basis of a program of insect trapping in the forest, that the perennial use of fenitrothion did not drastically

reduce the abundance and diversity of wild bees. Unfortunately, Varty and Carter (1974) did not have controls: they based their study on a comparison of two sites, one sprayed with fenitrothion for four years from 1969-1972, the other sprayed in the year of sampling, 1972. As both sites had been sprayed in the year of sampling, it is perhaps not surprising that their results were inconclusive. Their study relied upon counts of insects collected at window-traps, which are not effective for bees and give highly variable results for other insects. In another Canadian research project, Buckner (1975) indicated reductions in bumblebee populations of 50% in one study but none in another.

The preponderance of work on this topic, however, shows that, at least in New Brunswick, a reduction of pollinator populations as a result of fenitrothion spraying is the normal course of events. Much of the research concerns pollinators of lowbush blueberries (*Vaccinium angustifolium*). Kevan and Lear (1973), Kevan (1974, 1975b, 1975c, 1976, 1977a, 1977b) and Kevan and LaBerge (1979) noted that populations of blueberry pollinators were significantly lower (Kevan and Lear 1973; Kevan 1975b) in areas where fenitrothion was used than in areas where it had not been sprayed. Wood (1974) recognized the problem of the reduction (up to 90%) and lack of pollinators on blueberry fields in the vicinity of forest treated with fenitrothion. He published further evidence in support of those findings (Wood 1977, 1979, 1980c, 1980d, 1982a, 1982b). Rice, supplying evidence in the litigation between Bridges Brothers Limited and Forest Protection Limited in 1976, reported that from June 1-6 he found native bees in blueberry fields in populations between 800 and 1600 per sample, but after June 6 when the field was sprayed, the samples yielded less than 100 bees (New Brunswick Supreme Court 1976; Wood 1977; Kevan and LaBerge 1979).

Changes in pollinator abundance in other parts of the New Brunswick forest ecosystem were examined in several studies. Plowright et al. (1977, 1978a, 1978b), Plowright (1977, 1980), Plowright and Pendrel (1978), Plowright and Thaler (1979), Plowright and Rodd (1980), and Thomson et al. (1985) concluded that fenitrothion causes reductions in populations of bumblebees and other pollinators in treated forests. Bouchard (1981) arrived at similar conclusions for bumblebees in fenitrothion-treated areas in Quebec. Moreover, the demographic details revealed in those studies link, unequivocally, the observed reductions to applications of the insecticide. So far as bumblebees are concerned, for example, the late-emerging species (which are still in hibernation at the time of forest spraying) generally show no

reduction in population densities - by late summer, in fact, they tend to be more common in treated forest areas than in control plots. Plowright and Rodd (1980) suggested that may be the outcome of "ecological release," i.e. the late-emerging species benefit from the reduced densities of other earlier-emerging competitors.

2.3.2 Magnitude of Population Reduction

As indicated in the preceding section, in almost all reports on fenitrothion effects on pollinator populations it is concluded that substantial reductions occur. Reports indicating otherwise are either problematic (e.g. Varty and Carter 1974) or open to more than one interpretation (e.g. the studies by Buckner and co-workers).

The question now arises as to how large is "substantial"? What proportion of pollinating insects is generally killed or otherwise incapacitated by fenitrothion spraying? There is no single answer, because of differences in the questioners' frames of reference in space and time. For example, on the day following the application of an aerial fenitrothion spray, one could frequently search in the middle of a New Brunswick spray block without finding a single bumblebee queen (Plowright, personal observation). Or, even when bumblebees are present, their abundance may be reduced by more than an order of magnitude (e.g. the counts on *Clintonia borealis* in Plowright and Rodd (1980)). Further evidence for that sharp population reduction was obtained by nectar measurements in one of the principal bumblebee forage plants, *C. borealis* (Plowright and Rodd 1980). The results indicate that for the population of those insects which had emerged from hibernation before spraying, mortality is total. Near the edge of the same block, however, or even in the middle of the block a few days after spraying, at least a few live and healthy bumblebees could be seen. Furthermore, if one monitors the bumblebee population within a fenitrothion spray block throughout the summer, a gradual recovery (see following section) is noted, until by the time that goldenrod (*Solidago* spp.) is in bloom in early August, densities of bumblebees return to at least one-fifth of their normal levels (Plowright et al. 1978a).

Other complications attend the monitoring of population trends in ecosystems subjected to aerial applications of fenitrothion. Despite the best intentions of spray pilots and those who guide them, it is seldom the case that the insecticide is evenly distributed over the extent of a spray block. Inevitably, gaps occur and some areas are oversprayed. As

indicated by the exposure-cage experiments of Plowright and co-workers (Plowright et al. 1978a; Plowright and Rodd, 1980; Thomson et al. 1985), insects that happen to occur in such gaps may well escape the toxic impact of the insecticide. That, coupled with the fact that insects from outside the sprayed area are apt to migrate into the spray block (lured by the rich nectar and pollen rewards that accumulate as a result of reduced competition within the spray block), provides a situation of confusing spatial and temporal complexity. It is further compounded by the heterogeneity of the forest canopy and the ability of the pesticide to reach bees.

2.3.3 Recovery of Pollinator Populations

The recovery of pollinator populations is an issue fostering disagreement in the literature. In the documented cases involving fenitrothion, Wood (1979) concluded (from data gathered in a blueberry field that was sprayed only once) that recovery took place in one to three years (see also Wood 1982a). Kevan and LaBerge (1979) obtained similar results at that same sample site but wrote that there is a need for longer recovery periods for chronically affected blueberry fields. Wood (1982a) concluded:

Kevan (1977[c]) suggested that it may require as long as ten years before wild bee activity returns to normal. Wood (1979), on the other hand, showed full recovery after three years...

However, what Kevan (1977c) had actually stated was:

Preliminary estimates [of the recovery of pollinator populations] range from as little as 3 or 4 years to over 10 years, depending on the severity of the decimation and other factors.

Kevan and LaBerge (1979) documented five years of continued recovery in chronically affected fields. W. Bridges (pers. comm.) indicated that full recovery had taken place by about 1985 except in one of his company's blueberry fields in New Brunswick.

The perceived dispute regarding the time for population recovery is perhaps explained by the fact that the investigators chose different locations on which to base their estimates. In areas where fenitrothion spraying is repeated yearly, one does not expect the rapid recovery that characterizes isolated plots which receive only a single insecticide application (Plowright et al. 1978a; Kevan and

LaBerge 1979). Several other important issues emerge from the points made about the rate of pollinator recovery. Firstly, those insect species which are fairly mobile (such as bumblebees) are likely to show more rapid recolonization of sprayed areas than those which form nesting aggregations in the same place yearly (e.g. solitary bees). Also, in some groups of pollinating insects, ecological replacement by later-emerging species provides a rapid mechanism for partial, albeit highly variable, population recovery (Plowright and Rodd 1980). Finally, one must be careful to specify just what is meant by "complete recovery." Given that the attainment of pre-spray levels is likely to be an asymptotic process, the concept of "completeness" seems open to a range of different interpretations.

2.4 EFFECTS OF FENITROTHION ON PLANT FECUNDITY

2.4.1 Fenitrothion and Pollen

2.4.1.1 Pesticide toxicity to pollen: laboratory studies

Pollen has not been studied extensively with respect to its vulnerability to pesticide poisoning. Nevertheless, in about 30 studies published from 1975-1987, it is clear that pesticides and adjuvants generally reduce the viability or germinability of pollen grains, cause aberrations in their growth, cause aberrations during meioses and mitoses, reduce their rate of growth, or cause their death after germination. In studying the effects of 0.5% fenitrothion on pollen germination and pollen-tube growth in *Solanum melongena*, Regupathy et al. (1973) reported that the insecticide treatment significantly reduced pollen germination and pollen-tube growth (Figure 2.3). Kauer and Grover (1985) reported that pollen mother cells of *Hordeum vulgare* are adversely affected by organophosphorus insecticides, including fenitrothion, applied to seeds and to seedlings (Figure 2.4). They described a wide variety of chromosomal aberrations. It should be noted they in both studies the concentration of fenitrothion was much greater than that of forestry or agricultural formulations.

2.4.1.2 Toxicity to pollen and its effect on seed or fruit set

The information, scant as it is, suggests that there may be a significant increase in adversely affected pollen after fenitrothion use. While we feel that is unlikely (see below), we include a review of available

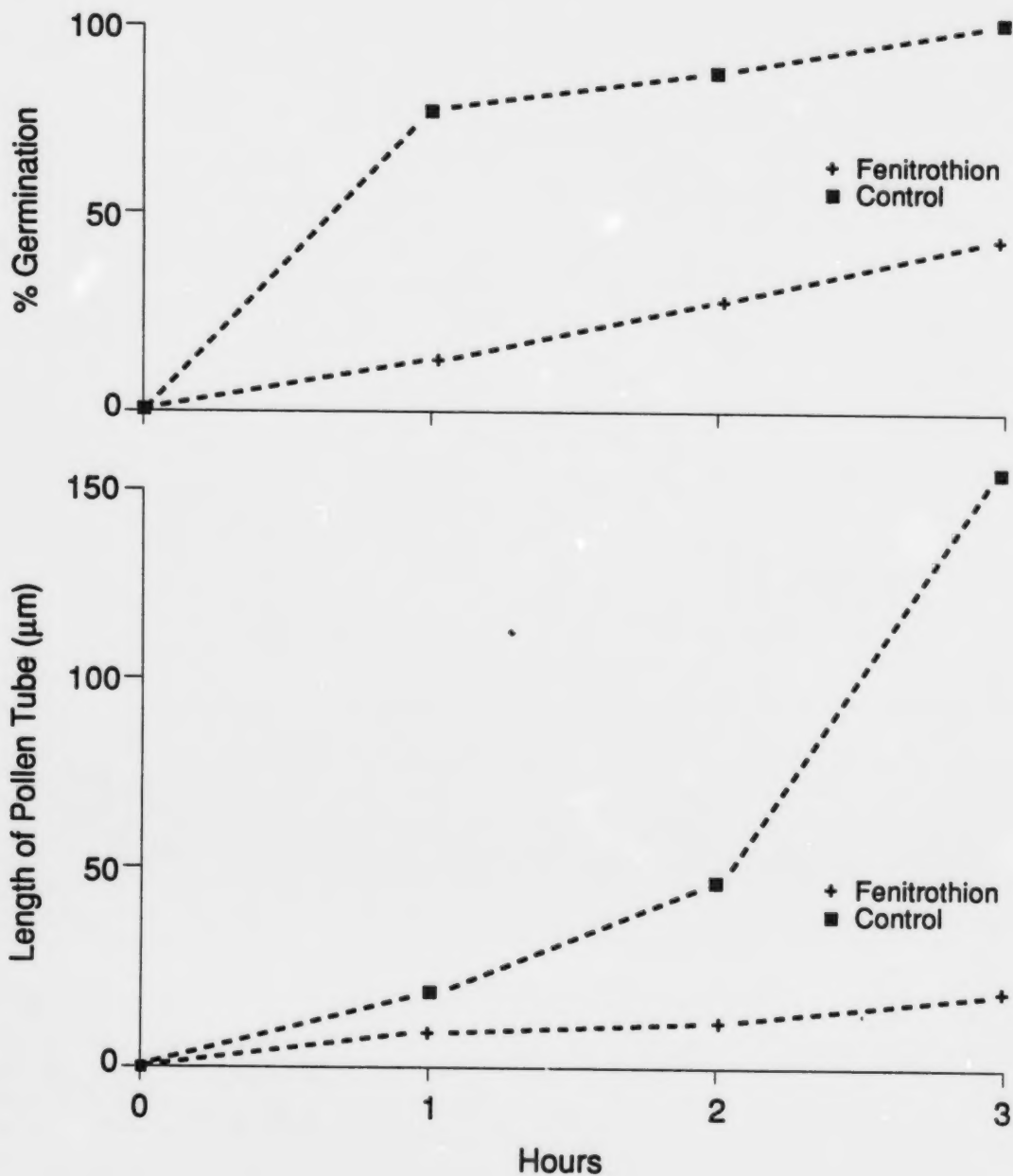


Figure 2.3 The percentage germination (upper) and rate of tube growth of pollen of *Solanum melongena* treated with fenitrothion (0.05%) (after Regupathy et al. 1973).

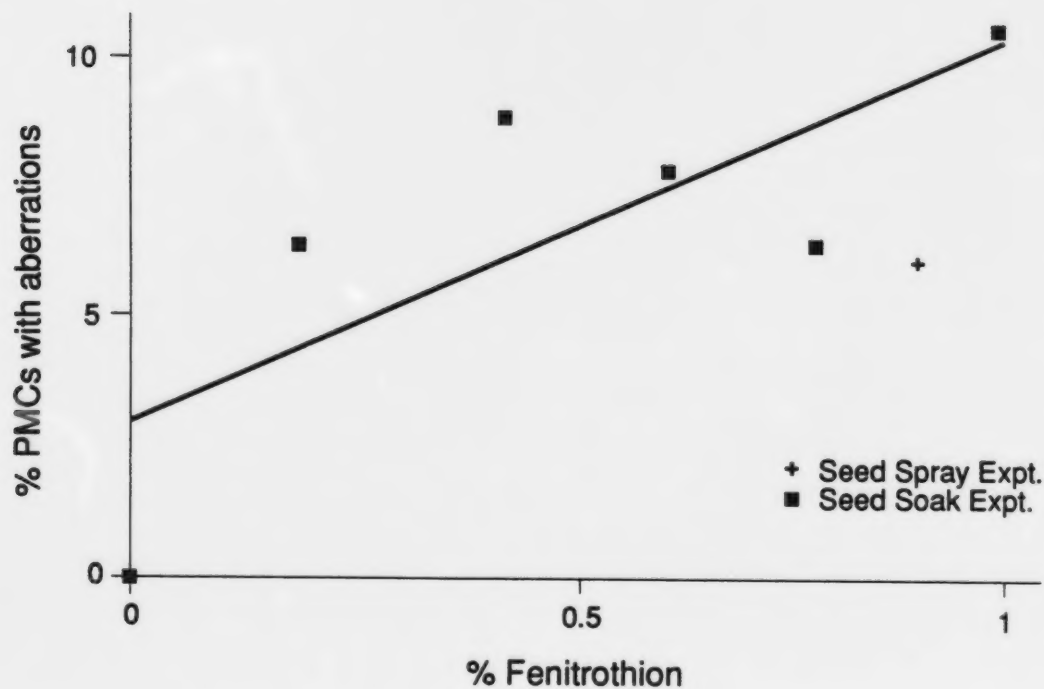


Figure 2.4 The effect of fenitrothion on the percentage of aberrations in pollen mother cells (PMCs) of *Hordeum vulgare* at two stages, seedlings (seed spray expt.) and seeds (seed soak expt.) (after Kauer and Grover 1985).

information for the sake of completeness. Only six studies known to us relate pesticide applications to pollination success as measured by seed set, none with fenitrothion. Lockhart (1967) studied the effects of several fungicides on pollen of lowbush blueberries and the number of seeds. He noted that fruit set was not reduced. Bristow and Shawa (1981) examined the effects of several fungicides on pollen germination and yield in cranberry, *V. macrocarpon*, finding some inhibited pollen germination as well as adversely affected yield. However, floral form is an important factor because similar results were not found in highbush blueberry, *V. corymbosum*, in which the pollen and stigma are protected in an urceolate, hanging flower (Bristow 1981). Paek and Cheong (1977) reported similar effects with some pesticides, noting that phosphamidon and difolatan caused reductions in the seed set of apples and pears. Gentile et al. (1973) found that applications of both carbaryl and tetrachlorvinphos caused malformed or missing kernels in corn, *Zea mays*. Debarr and Matthews (1985) applied azinphosmethyl, fenvalerate, and permethrin to *Pinus taeda* in bloom. They detected no adverse effects in any aspect of pollination or seed set that they examined.

Unfortunately, there is very little research on fenitrothion effects, and none on plants of the Canadian boreal region or of importance to Canadian agriculture or horticulture. Yet the experimental methods required are simple, at least for in vitro studies, in which pollen can be germinated in standard media in standard preparations (Stanley and Linskens 1974) and dosed with fenitrothion. We advise that research of that nature be undertaken even though, as mentioned above, we feel that field applications of fenitrothion could have only the most minimal, if any, effects on seed set through pollen toxicity. That is because the insecticide concentrations used in forest protection are much lower than those employed to show effects in the laboratory: individual pollen grains which come into contact with fenitrothion during a spray application would be vastly outnumbered by those that did not (a pollen grain is a very small target and the dosage rates used in forestry result in only two or three droplets impacting on a whole needle of balsam fir); and only pollen shed before or during the application would be at risk, leaving all other pollen presumably unaffected.

2.4.2 Reduction of Fruit Set Caused by Fenitrothion Applications

2.4.2.1 Magnitude and scope of effect

According to Kevan (1975b) and Wood (1977, 1982a), the suspicion that aerial fenitrothion spraying for

spruce budworm control might be causing reduced fruit production in forest plants first surfaced in 1970. Several blueberry growers, with fields located near fenitrothion-sprayed forests, reported unusually low pollinator activity and attributed their crop losses to the spray operation. In the first empirical investigation of that question (Buckner 1975), negative results were reported but, as noted by Plowright et al. (1978a), the choice of the study species, pin cherry (*Prunus pensylvanica*), in Buckner's investigation was inappropriate because it bloomed before spray time. Subsequent work by Thorpe (1979, 1980) supports the link between fenitrothion applications and reduced fruit set in blueberries. Later research by Plowright and co-workers (Plowright 1977; Thaler and Plowright 1980; Thomson et al. 1985) repeatedly demonstrated reductions of fecundity in susceptible plant species. In some cases, those have been as high as 78% (Thaler and Plowright 1980).

Although the strong association between fenitrothion spraying and reductions of fruit set in susceptible plant species may be taken as a firm indication that the causal pathway operates through effects on the insects that pollinate them, a skeptic might object that the evidence, being merely correlational, has not established such a link beyond reasonable doubt. In the most recent paper cited in the preceding paragraph (Thomson et al. 1985) the investigators attempted to cope with that objection. They generated a complex prediction based upon the differential susceptibility of the pollinating insects of two plant species (*Maianthemum canadense* and *Cornus canadensis*) to the two major insecticides, fenitrothion and aminocarb, used in New Brunswick in the forest-spray program at the time of the study. Because the flower visitors of *Maianthemum* (flies, solitary bees, and principally bumblebees) were more vulnerable to aminocarb than those of *Cornus*, Thomson et al. (1985) predicted that whereas both plant species might be expected to show a reduction in fruit production in fenitrothion-sprayed areas, only *Maianthemum* would be affected in locations treated with aminocarb. That prediction was borne out by the data and so supports the claim that reduction in fruit production by those forest herbs is indeed mediated through the impact of the insecticide on the pollinators of the plants.

The foregoing studies identified a set of criteria whereby the susceptibility of plants to the effects of forest spray operations on their insect pollinators can be assessed. For maximum susceptibility, the following conditions must be satisfied: the pollinators of the plant must be vulnerable to the insecticide in question; the blooming period of the plant must

coincide with the period in which the reduction of the pollinator populations is most intense (i.e. during the spray operations and the weeks immediately thereafter); and the breeding system of the plant must be such that insect visitation is necessary for fruit production. All the plant species for which an association between reduction in fruiting and fenitrothion spraying has been demonstrated (*Aralia nudicaulis*, *C. borealis*, *C. canadensis*, *Kalmia angustifolia*, *M. canadense*, and *Trifolium pratense*) satisfy the foregoing criteria. On the other hand, the conditions are not met by pin cherry, the plant used in Buckner's (1975) study, nor by the buttercup (*Ranunculus* sp.) chosen for a recent investigation in Newfoundland (Fudge et al. 1986-87).

The firm SOMER, in collaboration with the Service d'entomologie et de pathologie of the Quebec Ministry of Energy and Resources (SOMER 1985), conducted an especially thorough study of fenitrothion and aminocarb effects on four of the plant species mentioned above (*A. nudicaulis*, *C. borealis*, *C. canadensis*, and *M. canadense*). That investigation, which was characterized by commendably large sample sizes, found that sexual reproduction was depressed in all four species, and the effect persisted one year after spraying in the case of *A. nudicaulis* and *C. borealis*. Unfortunately, in every case, the study was unsuccessful in factoring out the impact of one insecticide from that of the other. The authors specifically implicated fenitrothion in the effects on *A. nudicaulis*, *C. borealis*, and *C. canadensis*.

Fudge et al. (1986-87) conducted more recent research on the buttercup (*Ranunculus* sp.) in Newfoundland and found that fenitrothion spraying did not affect plant fecundity. Since the buttercup, as mentioned above, does not satisfy the conditions for vulnerability to insecticide application, that result is not surprising. Several other boreal forest herbs do satisfy those conditions rather well (Barrett and Helenurm 1987; Helenurm and Barrett 1987). One, *Polygonatum pubescens*, has recently been evaluated for its usefulness as an indicator species (Barber 1987, 1988). Since the reports of SOMER (1985) and Fudge et al. (1986-87), however, there has been no further research comparing plant reproductive success between fenitrothion-sprayed and unsprayed areas.

The studies cited above emphasize the importance of taking into account the breeding systems of plants selected for monitoring environmental impacts on pollination. Until recently, the state of knowledge of the pollination requirements and breeding systems of Canadian plants, whether natural, introduced, or agricultural, has been woefully deficient. Recently published work on native boreal forest species

(Barrett and Helenurm 1987; Helenurm and Barrett 1987) improved the situation somewhat and some of our agricultural crops, especially fruits and forage crops, are understood, albeit mostly from research done elsewhere. But with a few exceptions (Brittain 1933; Wood 1965; Plowright and Hartling 1981; Boyle and Philogène 1983; Boyle-Makowski 1987; Kevan et al. 1984; Skepasts et al. 1987; Mohr and Jay 1988), the ecology of pollination has rarely been studied in crops grown in Canada. In general, it is assumed that what is applicable to the United States or northern Europe is applicable to Canada.

Looking beyond the mere fact of reductions in plant fecundity, the following questions should be asked. What is the effect of such reductions upon the population biology of the plants in question? Do the reductions have any secondary impact upon the functioning of the ecological communities of which the plants are a part?

2.4.2.2 Effect of reduced fecundity on plant population biology

Almost no work has been done on the effect of reduced fecundity caused by fenitrothion on the population biology of plants. Surveys in New Brunswick made during the summers of 1978 and 1979 produced inconclusive results (Plowright 1980). On the face of it, reduced fruit production should be reflected in diminished recruitment into the next generation or reduced dispersal of the plant. In practice, demonstrating that would be a task of formidable difficulty, because for most of the plant species in question the recruitment of seedlings is extremely low (Ashmun and Pitelka 1985; Cook 1985; T. J. Carleton, pers. comm.). Those species are very long-lived and reproduce primarily by vegetative means. Therefore, even if forest spraying reduced the establishment of new seedlings by a factor of four or five, the chance of showing it empirically, even with a small army of trained botanists, is highly remote. Despite that, we are of the opinion that the long-term consequences of reduced recruitment could be serious and certainly merit further study.

Apart from demographic changes, one might also expect shifts in the genetic composition of some plant populations as a result of reduced insect pollination. That expectation springs from the fact that many of the plant species in which fecundity is affected by fenitrothion spraying are at least partially self-compatible (Barrett and Helenurm 1987). In circumstances in which visits from their insect pollinators are few, such plants (e.g. *C. borealis*), commonly produce self-fertilized seed (Galen et al. 1985). The corollary, in terms of population genetics,

is that as self-fertilization is augmented, so is the genetic variance of the progeny reduced. In principle, that result could perhaps be demonstrated using appropriate techniques, such as electrophoretic allozyme marker analysis. Again, however, the cost of such an investigation would likely be very high.

2.4.2.3 Effect of reduced fecundity on community function

Kevan and Collins (1974) suggested the following chain of events. Assuming a reduction in seed and fruit production on the part of the plants whose insect pollinators were severely reduced by insecticide spraying, one might expect a response on the part of the organisms (birds, small mammals, and other insects for example) which normally depend on those seeds and fruit as part of their diet. The exact nature of such a response can only be a matter for speculation, but it is certainly quite possible that some of those frugivorous organisms might turn their attention, as suggested by Kevan and Collins (1974), to commercial blueberry fields as a substitute for their natural food sources. That, again, is a research area requiring an enormous effort, because detecting the expected dietary shift in the heterogeneous complexity of natural forest systems requires huge sample sizes.

2.5 ECONOMIC IMPACT

The economic impact of pesticides on pollinators has been assessed mostly on the basis of losses of honeybees. Various indemnity programs have been instituted in some places. No information relating to fenitrothion, specifically to losses in honeybees, is readily available. It takes time and effort to uncover existing records. Johansen (1977) provided some idea of the cost of poisoned pollinators by citing the example of alkali bees (*Nomia melanderi*) as pollinators of alfalfa in Washington. Estimated losses in seed production were \$287,000 in 1973, and the bee populations regained only 25% of their initial population strength two years later.

The value of pollination to agriculture exceeds the value of bees and bee products by orders of magnitude (Levin 1983; Winston and Scott 1984), so the demise of pollinators may be expected to have serious ramifications. Kevan and Oppermann (1980), in augmenting data in Kevan (1977c) and Kevan and LaBerge (1979), and answering the criticisms of Wood et al. (1979), claimed that from 1970-1977 the blueberry harvest in New Brunswick was depressed about 0.67 million kg below its potential. The year 1970 coincides with the arousal of the suspicions of

blueberry growers that the insecticide was decimating native pollinators.

No statistical analyses were made for other crops in New Brunswick over the same period addressed in the study on blueberry production. There is no subsequent study (incorporating the most recent decade) to determine if the return of the pollinators and changes in pollination management have been accompanied by increases in blueberry production relative to that in neighbouring Nova Scotia or Maine. Because data (Table 2.2) are now available from Statistics Canada on blueberry production from New Brunswick and Nova Scotia up to 1987, a statistical analysis to complement Kevan and Oppermann's (1980) study has been carried out. Using the general linear modelling program GLIM (Version 3.77, Royal Statistical Society of London), and treating province, year and incidence of spraying as factors in a gamma-distributed error model, we obtained a significant ($P < 0.05$) average annual depression in total production in New Brunswick of 665 t over the period (1971-1979) when fenitrothion was considered to have affected crop production in that province. That agrees with Kevan and Oppermann's (1980) estimate of 0.67 million kg from the earlier part of the data set (which did not include the post-fenitrothion effect period).

It is clear that the economic impact of pesticide use goes far beyond the loss of bees. In considering the use of fenitrothion, the crops which may be adversely affected must be considered in the benefit/cost ratio as an important component of cost. In recent years buffer zones, where insecticide spraying is prohibited, have been in place around blueberry fields (Sexsmith 1987).

As a postscript to this section, we point out that the reason why data from blueberry fields in New Brunswick are so relevant is that such fields are an integral and characteristic part of the forest environment. Most are abandoned farm fields more or less surrounded by native forest. The blueberries are themselves native plants normally present in the forest, especially in clearings, which are managed (no ploughing, occasional herbicide and insecticide applications, and regular burning to prune and control invading woody plants). The fields are patches of regenerating forest held at an early and exaggerated seral stage. The wild bee flower-visitors of blueberries, which number 192 species (Finnamore and Neary 1978) are also nearly all native and can be found in the forest proper. Thus, although blueberry fields lack an arboreal canopy, they represent an example of a pollination association which is an essential part of the natural regeneration of the

forest ecosystem following fire or other physical or biologically caused catastrophe.

2.6 LEGAL ISSUES

The decision for the plaintiff, Bridges Brothers Limited (blueberry farmers), and against Forest Protection Limited (fenitrothion sprayers) by the New Brunswick Supreme Court (1976) was a

landmark in Canadian environmental law. A small settlement resulted because of weak, highly conjectural, and usually untested arguments on both sides.

No similar court action has subsequently been taken in Canada. However, in a wider context, numerous legal actions have been taken over the killing of honeybees throughout the western world and, in many places, laws exist to protect honeybees and beekeepers. Crane and Walker (1983) provide a

Table 2.2 Blueberry crop yields (in tonnes) from New Brunswick and Nova Scotia from 1951 to 1987 (Statistics Canada). The years 1970-1977 were affected by fenitrothion spraying in New Brunswick but not in Nova Scotia.

<i>Year</i>	<i>New Brunswick</i>	<i>Nova Scotia</i>
1952	2000	1050
1953	640	562
1954	1750	2563
1955	1500	1400
1956	2050	2010
1957	1650	2400
1958	1250	1500
1959	1750	2600
1960	1750	2700
1961	2250	2850
1962	2000	3700
1963	2000	3500
1964	1500	2550
1965	1250	3500
1966	3500	3800
1967	3500	5850
1968	750	1050
1969	2750	4640
1970	750	4150
1971	2400	3600
1972	1890	5000
1973	2325	5100
1974	1392	3850
1975	1918	5050
1976	1880	3515
1977	1652	4175
1978	1850	5900
1979	2139	5350
1980	1950	4324
1981	2550	6492
1982	3850	7228
1983	4235	9926
1984	2183	7733
1985	2600	9660
1986	3000	8255
1987	3500	6796

summary of legislation from around the world. Given the similarity of jurisprudence in Canada and Great Britain, the case of *Tutton vs. A. D. Walter Ltd.* in England in 1984 is important (WLR 1985). In that case, the judge ruled for the plaintiffs, beekeepers, whose bees had been killed by a pesticide application to oil seed rape, in flower, in fields near their apiaries. Significantly, the reasoning of the court was not only the careless application of the spray but also the lack of opportunity given beekeepers to take measures to protect their bees. Both parties have responsibility under the "common duty of care judged by the 'neighbour' criterion" (Kevan 1987).

It is important that the issues raised above be thoroughly understood by all concerned parties as well as the public.

2.7 CRITIQUE

The fact that fenitrothion is one of the best researched insecticides from the standpoint of its effects on the environment is both a blessing and a curse. It is a blessing in that a large body of information exists in the primary literature (i.e. research reports and reviews published in refereed scientific journals). It is a curse because around the ecotoxicology of fenitrothion, a large amount of "gray" literature exists in the form of internal reports and reviews. Some parts of that literature contain primary and valuable scientific findings reported by qualified and competent scientists (though all but buried to the scientific community at large). Other parts contain repetitive reiterations of the internal reports. Still other parts, and those are generally the worst in quality, consist of second-hand reporting of any of the above-mentioned material, or reports of poorly conceived research designed and conducted by unqualified investigators or both. A major problem is the internal recycling of such information, together with the ignoring, or at least short-shrifting, of externally generated and published results of research. Given the above, it is not surprising that some reports (e.g. Poole et al. 1981), even though compiled by highly educated personnel for the highest levels of government to aid in decision-making, are wanting in terms of informed opinion. Others, such as LAVALIN (1984), are so caught up in the process of recycling the gray literature that they do little more than create bibliographic confusion.

The uneven quality of the research on the ecotoxicology of fenitrothion has, of course, been noted before. In April, 1977, after a symposium on the long-term effects of fenitrothion use in forest

ecosystems (sponsored by the NRCC in association with the Federal Interdepartmental Committee on Pesticides), NRCC's Associate Committee on Scientific Criteria for Environmental Quality convened a panel chaired by one of Canada's most highly respected senior scientists, P. D. McTaggart-Cowan. The terms of reference of the panel were to prepare a report on "where we now stand scientifically and what needs to be done." As indicated earlier in the present review, the resulting document (NRCC 1977b) was sharply critical of the standard of scientific work in the area. The Chairman's Critique, which precedes the panel's report, is unusually scathing (NRCC 1977b):

With some outstanding exceptions, most of the scientific work done to date lacks rigour, planning, and control, and one wonders whether there has been more concern with appearing busy than with shedding light on the risks and benefits.

Looking back, and with the advantage of hindsight, both insightful commentary and some distinctly unhelpful hand-waving can be recognized in the report of McTaggart-Cowan's panel. The panel advocates the carrying out of "key experiments," and it stresses the need for "an imaginative, innovative, and truly scientific approach." But just what are those key experiments, and how would an imaginative and innovative approach differ from what has been done during the past decade and a half? On the latter point, the panel's report is quite inscrutable, but it does at least recommend a key experiment or two:

Insects which appear to be significantly affected by fenitrothion applications include pollinating Hymenoptera... However, the key experiments to assess the impact due to the removal of this component from the forest ecosystem have not been performed.

In one sense, that must stand as one of the more fatuous recommendations relating to the environmental monitoring of forest spray operations made to date. Those who are making a living as insect ecologists do not lightly contemplate removing an entire component from a forest ecosystem, even for the purposes of ecotoxicological experimentation.

But in another sense, the panel's report spoke with good sense to issues which need addressing today as much as they did in 1977. Firstly, it is still the case

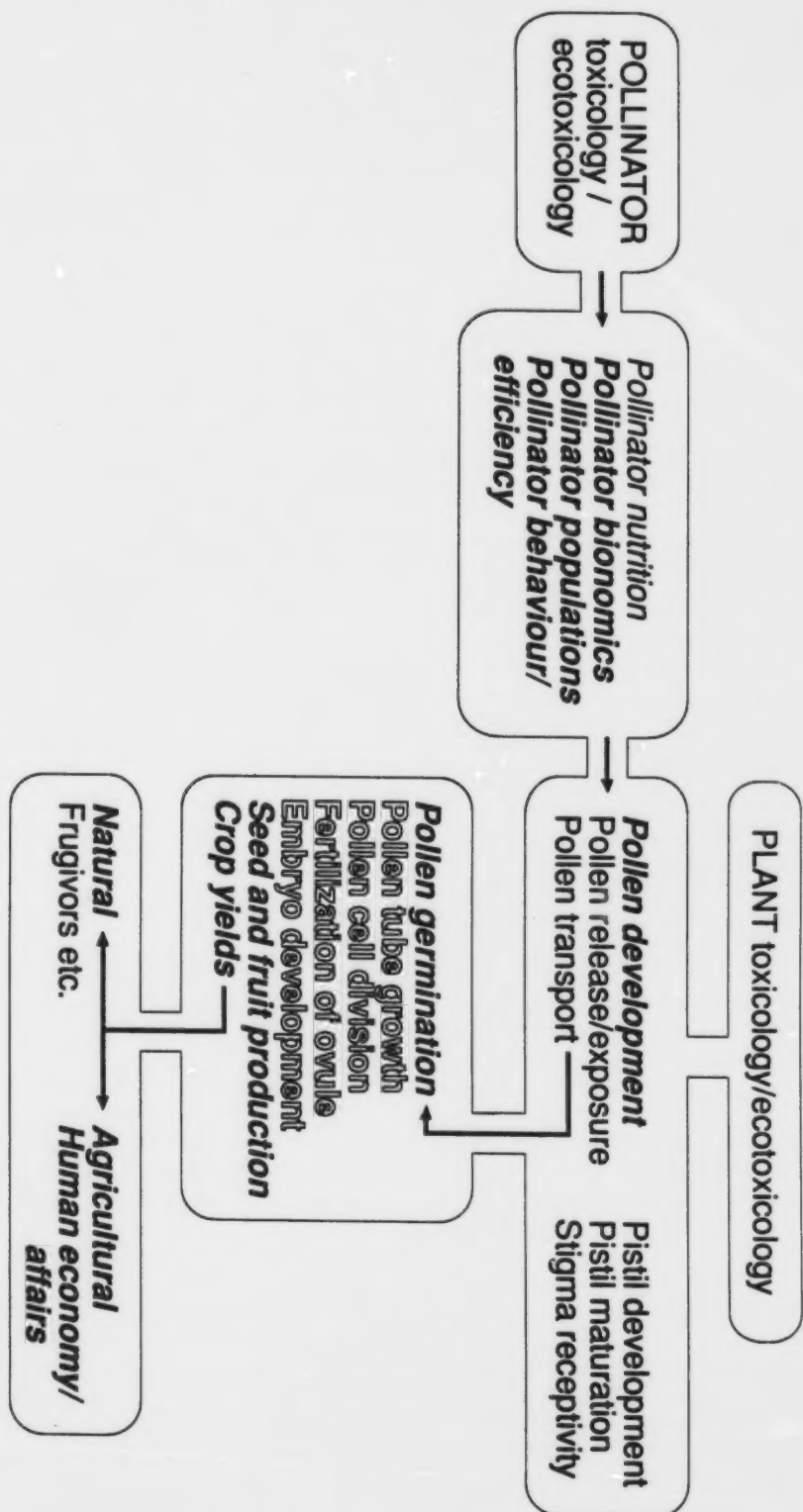


Figure 2.5 **Components of pollination directly affected by fenitrothion (bold italic)**
indirectly affected (italic) and presumably affected (outline)
together with processes known to be directly affected →

that the effects of insecticides are little understood from the point of view of the systems upon which they impinge. Whether taken from an anthropocentric or naturalistic perspective, the scientific appreciation of the interactive network of causes and effects is woefully lacking. Clearly, a synthetic systems approach, by which the components are identified, is needed. An attempt was made to identify at least some of the components and to note huge gaps of knowledge that exist regarding those listed in Figure 2.5, even though fenitrothion is one of the most studied insecticides. Secondly, in order for the work just described to be carried out effectively, some overall coordination, on a national level, seems essential.

2.8 INFORMATION DEFICIENCIES

Information needs are grouped into three categories. The first category concerns the ecotoxicology and biology of pollination. It describes the projects and general approaches that should be taken and why they should be taken. The second category addresses the socio-economic and legal aspects of the impact of pesticides on pollination. It involves the need for fuller understanding of the benefit/cost equation as well as the transfer of accurate legal information. The third category addresses research methodology and administration. It stems from concern over the amount of effort misdirected through basic ignorance.

2.8.1 Pollination/Pollinator Ecotoxicology and Biology

There is a need for further research on fenitrothion effects on pollinator populations, not for its own sake but as a background to analyses of the more complex web of interactions.

There is a need to document the diversity of the reproductive systems of plants likely to be affected by fenitrothion applications to forests as a basic background to further studies.

There is a need to investigate fenitrothion effects on reproductive physiology and sporogenesis in plants. Such studies should be designed to elucidate the nature of subtle disruptions of plant reproduction via disruption of the normal course of mitoses, meioses, cell division, cell growth, and fertilization. Virtually no relevant information is currently available for Canadian plants.

There is a need for rigorous studies of fenitrothion effects on honeybees. The spatial and temporal dynamics of the demography and population biology (division of labour, recruitment, and foraging activity, etc.) of honeybees' colonies need to be understood from the points of view of scientists and beekeepers. Investigations should be designed to provide a firm and an unequivocal basis for the assessment of damages, both ecological and financial. New approaches need to be taken in combination with the standardization of methodologies pioneered by the International Commission for Bee Botany (1980).

There is a need for scientific research on the effects of sublethal doses of fenitrothion on pollinators. It must be a part of ecotoxicological research on pollination, apiculture, and pollinators because the subtle effects of fenitrothion may, in aggregate, be more disruptive of pollinator activity than direct mortality.

There is a need to understand the spatial and temporal dynamics of pollinator populations and of plant fecundity in both natural and agricultural ecosystems which have, and have not, been affected by fenitrothion. Appropriate studies could elucidate the importance of non-linearities in the interactive processes associated with pollination. Some effects may be synergistic, compounding, or buffered. They need investigation through an integrative systems approach whereby components are itemized, modeled appropriately, and then recombined in holistic, dynamic models.

2.8.2 Socio-Economic and Legal

There is a need to assess the economic costs of fenitrothion spraying not only in terms of tangible losses of honeybees and reduced harvests of commercial crops but also of presently intangible costs such as wild bee mortality and other ecological side effects. Both the direct, and ultimately the intangible, costs should be entered into the benefit/cost ledger.

2.8.3 Research Methodology and Administration

Competent and qualified scientists should be given adequate resources to carry out the research summarized in the foregoing recommendations. A standard provision in future work should be the involvement of recognized experts in pollination and pollinator biology not only in the planning and review of the research but also in the interpretation of its results.

2.9 LITERATURE CITED

- Abdel-Wahab, A.M., F.M. Khalil, and M.H. Hussein. 1973. Toxicity of insecticides to honeybees in laboratory and field (Hymenoptera: Apidae). Bull. Entomol. Soc. Egypt 7: 83-89.
- Anderson, L.D. and E.L. Atkins. 1968. Pesticide usage in relation to beekeepers. Ann. Review Entomol. 13: 213-238.
- Anderson, L.D., E.L. Atkins, H. Nakakihara, and E.A. Greywood. 1971. Toxicity of pesticides and other agricultural chemicals to honey bees: field study. Univ. Calif. Agric. Ext. Publ. AXT-251. 8p.
- Ashmun, J.W. and L.F. Pitelka. 1985. Population biology of *Clintonia borealis*. II. Survival and growth of transplanted ramets in different environments. J. Ecol. 73: 185-198.
- Atkins, E.L. 1975. Injury to honey bees by poisoning. In Dadant & Sons (eds.). The Hive and the Honey Bee. Dadant & Sons, Hamilton, Illinois. p. 663-696.
- Atkins, E.L., L.D. Anderson, H. Nakakihara, and E.A. Greywood. 1970. Toxicity of pesticides and other agricultural chemicals to honey bees: laboratory studies. Univ. Calif. Agric. Ext. Publ. M-16. 25 p.
- Atkins, E.L., E.A. Greywood, and R.L. MacDonald. 1973. Toxicity of pesticides and other agricultural chemicals to honey bees: laboratory studies. Univ. Calif. Agric. Ext. Publ. M-16 (Revised).
- Bacilek, J. 1982. Model for rapid field evaluation of the toxicity of pesticides to honeybee colonies. J. Apic. Res. 21: 57-61.
- Bai, A.R.K. and C.C. Reddy. 1977a. Laboratory toxicity of some insecticides to *Apis cerana indica*. J. Apic. Res. 16: 161-162.
- Bai, A.R.K. and C.C. Reddy. 1977b. Inhibition of acetylcholinesterase as a criterion to determine the degree of insecticide poisoning in *Apis cerana indica*. J. Apic. Res. 16: 112-114.
- Barber, K.N. 1987. Developing monitoring techniques to measure insecticidal impacts on plant fecundity and solitary bees. Ontario Pesticides Advisory Committee, Ontario Ministry of the Environment. (Abstract presented at 1987 Annual Meeting).
- Barber, K.N. 1988. Developing an implementation protocol for fruit-set of the forest wildflower *Polygonatum pubescens* in an insecticide impact monitoring program. Ontario Pesticides Advisory Committee, Ontario Ministry of the Environment. (Abstract presented at 1988 Annual Meeting).
- Barrett, S.C.H. and K. Helenurm. 1987. The reproductive biology of boreal forest herbs. I. Breeding systems and pollination. Can. J. Bot. 65: 2036-2046.
- Barth, F.G. 1985. Insects and Flowers: The Biology of a Partnership. Princeton University Press, Princeton, N.J. 297 p.
- Benedek, P., J. Komlodi, G. Herczeg, and G. Dely. 1976. Preliminary experiments on seed lucerne pest control harmless to bees: toxicity of field weathered insecticide residues to honeybees [in Hungarian]. Novenytermeles 25: 129-140.
- Bouchard, Y. 1981. Impacte des pulvérisations aériennes de fénitrothion et d'aminocarbe sur les populations de bourdons au Québec. Unpublished Report, Dépt. Biologie, Univ. Laval, Québec. 42 p.
- Boyle, R.M.D. and B.J.R. Philogène. 1983. The native pollinators of an apple orchard: variations and significance. J. Hort. Sci. 58: 355-363.
- Boyle-Makowski, R.M.D. 1987. The importance of native pollinators in cultivated orchards: their abundance and activities in relation to weather conditions. Proc. Ent. Soc. Ont. 118: 125-141.
- Bristow, P.R. 1981. Effect of triforine on pollen germination and fruit set in highbush blueberry. Plant Dis. 65: 350-353.
- Bristow, P.R. and A.Y. Shawa. 1981. The influence of fungicides on pollen germination and yield in cranberry. J. Amer. Soc. Hort. Sci. 106: 290-292.

- Brittain, W.H. 1933. Apple pollination studies in the Annapolis Valley, N.S., Canada. 1928-1932. Dominion of Canada. Department of Agriculture Bulletin 162: 1-198.
- Buckner, C.H. 1974. The biological side-effects of fenitrothion in forest ecosystems. Chem. Control Res. Inst., Inf. Rep. CC-X-67, Can. For. Serv. 29 p.
- Buckner, C.H. 1975. Supporting documents supplied to the expert panel on fenitrothion. NRCC Publ. No. NRCC 14104, Ottawa.
- Buckner, C.H., T.A. Gochbauer, and B.B. McLeod. 1975. The impact of aerial spraying of insecticides on bees. In M.L. Prebble (ed.). Aerial control of forest insects in Canada. Dept. Environment, Ottawa. p. 276-279.
- Buckner, C.H. and S. Sarrazin. 1975. Studies on the environmental impact of the 1974 spruce budworm control operations in Quebec. Chem. Control Res. Inst. Inf. Rep. CC-X-93, Can. For. Serv. 196 p.
- Buckner, C.H. and B.B. McLeod. 1977. Ecological impact studies of experimental and operational spruce budworm control programs on selected non-target organisms in Quebec, 1976. For. Pest Manage. Inst., Inf. Rep. CC-X-137, Can. For. Serv. 81 p.
- Capizzi, J. et al. (eds.). 1979. Pacific Northwest Insect Control Handbook. Handbook of the Extension Service of Oregon State University, Washington State University, and University of Idaho. 264 p.
- Cook, R.E. 1985. Growth and development in clonal plant populations. In J.B.C. Jackson et al. (eds.). Population Biology and Evolution of Clonal Organisms. Yale University Press, New Haven, and London. p. 259-296.
- Cox, R.R. and W.T. Wilson. 1984. Effects of permethrin on the behaviour of individually tagged bees. *Apis mellifera* Hymenoptera: Apidae. Environ. Entomol. 13: 375-378.
- Cox, R.R. and W.T. Wilson. 1987. The behaviour of insecticide-exposed honey bees. Amer. Bee J. 127: 118-119.
- Crane, E. and P. Walker. 1983. The Impact of Pest Management Bees and Pollination. International Bee Research Association and Tropical Development and Research Institute, London. ix + 129 p., + 73 p., + 9 p.
- Crepet, W.I. 1983. The role of insect pollination in the evolution of the Angiosperms. In L. Real (ed.). Pollination Biology. Academic Press, Orlando. p. 31-48.
- Debarr, G.L. and F.R. Matthews. 1985. Insecticide applications during pollination period do not adversely affect seed-yields of loblolly pine, *Pinus taeda*. South. J. Appl. For. 9: 240-243.
- Edinburgh School of Agriculture. 1982. Annual Report 1981. Edinburgh School of Agriculture, Edinburgh. 214 p.
- FAO & WHO. 1970. 1969 Evaluations of some pesticide residues in food. Food and Agriculture Organization and World Health Organization of the United Nations, Rome.
- Faegri, K. and L. van der Pijl. 1971. The Principles of Pollination Ecology. Pergamon Press, Oxford. 291 p.
- Finnamore, B. and M.E. Neary. 1978. Blueberry pollinators of Nova Scotia, with a check list of the blueberry pollinators of eastern Canada and northeastern United States. Ann. Entomol. Soc. Que. 23: 168-181.
- Free, J.B. 1970. Insect Pollination of Crops. Academic Press, London. 544 p.
- Fudge, S.M., J. Pratt, V. Anstey, S. Meron, B. Hicks, and K. Gilkinson. 1986-87. A study to monitor the effects of aerially applied fenitrothion on the activity of Newfoundland pollinating insects. Prepared for Newfoundland and Labrador Department of Forest Resources and Lands, Corner Brook, Newfoundland. Fudge, Lane and Associates Limited, St. John's, Newfoundland. 11 p. + Tables.
- Galen, C. and R.C. Plowright. 1985. The effects of nectar level and flower development on pollen carry-over in inflorescences of fireweed (*Epilobium angustifolium*) (Onagraceae). Can. J. Bot. 63: 488-491.

- Galen C., R.C. Plowright, and J.D. Thomson. 1985. Floral biology and regulation of seed set and seed size in the lily *Clintonia borealis* (Ait.) Raf. Amer. J. Bot. 72: 1544-1552.
- Gentile, A.G., A.W. Vaughan, S.M. Richman, and A.T. Eaton. 1973. Corn pollen germination and tube elongation inhibited or reduced by commercial and experimental formulations of pesticides and adjuvants. Environ. Entomol. 2: 473-476.
- Gorecki, K. 1973. Harmful effects of insecticides used in Poland on honey bee *Apis mellifera* Hymenoptera: Apidae. Polskie Pismo Entomologiczne 43: 201-210 [Bioabstracts B570; 1545].
- Grant, W.F. 1984. Plant Biosystematics. Academic Press, Toronto. 674 p.
- Hameed, S.F., R.L. Adlakha, and S.P. Gaimzo. 1973. Relative toxicity of some insecticides to the workers of *Apis mellifera*. Madras Agr. J. 60: 552-556.
- Heinrich, B. 1979. Majoring and minoring by foraging bumble-bees. *B. vagans*: An experimental study. Ecology 60: 245-255.
- Helenurm, K. and S.C.H. Barrett. 1987. The reproductive biology of boreal forest herbs. II. Phenology of flowering and fruiting. Can. J. Bot. 65: 2047-2056.
- Ibrahim, S.H., A.M. Madkour, and H.A. Selim. 1971. Residual effect of some new pesticides to honeybees. Agric. Res. Rev. 49: 94-98.
- Indian Standards Institution. 1979. Sectional List of Indian Standards: 1. Agricultural and Food Products. Indian Standards Institution, Pune, India. 26 p.
- International Commission for Bee Botany. 1980. Report of the meeting of the symposium on the harmonization of methods for testing the toxicity of pesticides to bees, 23-25 September 1980, Wageningen, The Netherlands. International Commission for Bee Botany. 13 p. + Appendices.
- Johansen, C.A. 1977. Pesticides and pollination. Ann. Rev. Entomol. 22: 177-192.
- Johansen, C.A. 1979. Honeybee poisoning by chemicals: signs, contributing factors, current problems and prevention. Bee World 60: 109-127.
- Johansen, C.A. and E.L. Atkins. 1978. Bee Toxicology: Sumithion 8EC: Test No. 1-59-P-78. Unpublished study received by EPA Feb 6, 1979 under 476-EX-80; prepared by Washington State Univ., Dept. of Entomology and Univ. California-Riverside, Dept. of Entomology. Submitted by Stauffer Chemical Co., Richmond, CA; CDL: 097792-C.
- Johansen, C.A. and J. Eves. 1967. Toxicity of insecticides to the alkali bee and the alfalfa leafcutting bee. Wash. Agric. Exp. Stn. Circ. 475: 1-15.
- Johansen, C.A. and D. Mayer. 1976. Alkali bees: their biology and management for alfalfa seed production in the Pacific Northwest. Pacific Northwest Extension Publication 155: 1-19.
- Kauer, P. and I.S. Grover. 1985. Cytological effects of some organophosphate pesticides. 2. Meiotic effects. Cytologia (Tokyo) 50: 199-211.
- Kevan, P.G. 1974. Effects of the 1973 forest spray programme on the wild bee pollinators of lowbush blueberries in southern New Brunswick. (Preliminary report). Unpublished Report to Bridges Bros., Ltd., Commercial Blueberry Farms, St. Stephen, N.B. 22 p.
- Kevan, P.G. 1975a. Pollination and Environmental Conservation. Environ. Conserv. 2: 293-298.
- Kevan, P.G. 1975b. Forest application of the insecticide fenitrothion and its effect on the wild bee pollinators (Hymenoptera: Apoidea) of lowbush blueberries (*Vaccinium* spp.) in southern New Brunswick, Canada. Biol. Conserv. 7: 301-309.
- Kevan, P.G. 1975c. Effects of the 1974 forest spray programme on the wild bee pollinators of lowbush blueberries in southern New Brunswick. (Preliminary Report.) Unpublished Report to Bridges Bros., Ltd., Commercial Blueberry Farms, St. Stephen, N.B. 23 p.

- Kevan, P.G. 1976. Effects of the 1975 forest spray programme on the wild bee pollinators of lowbush blueberries in southern New Brunswick. (Preliminary Report.) Unpublished Report to Bridges Bros., Ltd., Commercial Blueberry Farms, St. Stephen, N.B. 21 p.
- Kevan, P.G. 1977a. Results of pollinator sampling on lowbush blueberry fields in southern New Brunswick in 1976: monitoring the long-term effects of the 1970-1975 forest spray programme. Unpublished Report to Bridges Bros., Ltd., Commercial Blueberry Farms, St. Stephen, N.B. 28 p.
- Kevan, P.G. 1977b. Wild bee pollinators of lowbush blueberries as affected by the forest spray program of 1977 in southern New Brunswick. Unpublished Report to Bridges Bros., Ltd., Commercial Blueberry Farms, St. Stephen, N.B. 28 p.
- Kevan, P.G. 1977c. Blueberry crops in Nova Scotia and New Brunswick. Pesticides and crop reduction. Can. J. Agric. Econ. 25: 61-64.
- Kevan, P.G. 1987. Pesticides and Bees: A legal case from England with relevance to Canada. Can. Beekeeping 13 (9): 208.
- Kevan, P.G. and H.G. Baker. 1983. Insects as flower visitors and pollinators. Ann. Rev. Entomol. 28: 407-453.
- Kevan, P.G. and M. Collins. 1974. Bees, blueberries, birds, and budworms. Osprey (Newfoundland Natural History Society Newsletter) 5: 54-72.
- Kevan, P.G. and W.E. LaBerge. 1979. Demise and recovery of native pollinator populations through pesticide use and some economic implications. Proceedings of the 4th International Symposium on Pollination, Maryland Agric. Exp. Stn. Spec. Misc. Publ. 1: 489-508.
- Kevan, P.G. and A. Lear. 1973. The effect of the spruce budworm spray program on pollinator abundance on blueberry fields in southern New Brunswick during the blueberry bloom in early June, 1972. Unpublished Report to Bridges Bros., Ltd., Commercial Blueberry Farms, St. Stephen, N.B. 28 p.
- Kevan, P.G. and E.B. Oppermann. 1980. Blueberry production in New Brunswick, Nova Scotia, and Maine. A reply to Wood et al. Can. J. Agric. Econ. 28: 81-84.
- Kevan, P.G., R.M. Gadawski, S.D. Kevan, and S.E. Gadawski. 1984. Pollination of cranberries, *Vaccinium macrocarpon*, on cultivated marshes in Ontario. Proc. Entomol. Soc. Ont. 114: 45-53.
- Korpela, S. and U. Tulisalo. 1974. The residual contact toxicity to honey bees of eight organophosphorus insecticides on rape flowers tested with a simple laboratory method. Ann. Entomol. Fenn. 40: 1-9.
- LAVALIN. 1984. Évaluation socio-environnementale et économique. Programme de pulvérisations contre la tordeuse des bourgeons de l'épinette. Annexe 1. Toxicologie du fenitrothion et de l'aminocarbe. Dossier No. 35660 submitted by André Marsam et Associés to the Ministère de l'Energie et des Ressources, Québec.
- Levin, M.D. 1983. Value of bee pollination to U.S. agriculture. Bull. Entomol. Soc. Amer. 29: 50-51.
- Lockhart, C.L. 1967. Effect of fungicides on germination of lowbush blueberry pollen and number of seeds per berry. Can. Plant Dis. Surv. 47: 72-73.
- McGregor, S.E. 1976. Insect Pollination of Cultivated Crop Plants. Handbook No. 496, Agricultural Research Service, United States Department of Agriculture, Washington, D.C. 411 p.
- McLeod, B.B. and K.L. Mortensen. 1975. Honey Bees. In C.H. Buckner and R. Sarrazin (eds.). Studies of the Environmental Impact of the 1974 Spruce Budworm Control Operation in Quebec. Chem. Control Res. Inst. Inf. Rep. CC-X-93, Can. For. Serv.
- Mikolajczak, B. and M. Bohosiewicz. 1971. Laboratory diagnosis of bee poisoning by methyl parathion and fenitrothion. Med. Weter. 27: 459-460.
- Mohr, N.A. and S.C. Jay. 1988. The foraging behaviour of honey bees on canola (*Brassica campestris* L. and *Brassica napus* L.). J. Apic. Res. In Press.

- National Research Council Canada. 1975. Fenitrothion: The effects of its use on environmental quality and its chemistry. NRCC Publ. No. NRCC 14104, Ottawa. 162 p.
- National Research Council Canada. 1977a. Proceedings of a symposium on fenitrothion: the long-term effects of its use in forest ecosystems. NRCC Publ. No. NRCC 16073, Ottawa. 628 p.
- National Research Council Canada. 1977b. Fenitrothion: the long-term effects of its use in forest ecosystems. Current status. NRCC Publ. No. NRCC 15389, Ottawa. 18 p.
- National Research Council Canada. 1981. Pesticide-Pollinator Interactions. NRCC Publ. No. NRCC 18471, Ottawa. 190 p.
- New Brunswick Supreme Court. 1976. Between: Bridges Brothers Limited, plaintiff, and Forest Protection Limited, defendant. Queen's Bench Division, Fredericton. 59 p.
- Okada, I. and H. Hoshiba. 1970. A laboratory experiment on some insecticides to honeybee. Bull. Fac. Agric. Tamagawa Univ. 10: 79-85.
- Paek, K-Y. and S.T. Cheong. 1977. Effects of pesticides on pollen germination and fruit set in apple and pear cultivators during blooming periods. J. Korean Soc. Hort. Sci. 18: 125-136.
- Plowright, R.C. 1977. The effect of fenitrothion on forest pollinators in New Brunswick. Proceedings of a Symposium on Fenitrothion. NRCC Publ. No. NRCC 16073, Ottawa. p. 335-341.
- Plowright, R.C. 1980. Annual and seasonal trends in bumblebee abundance and flowering plant reproduction, associated with aerial spray operations. In I.W. Varty (ed.). Environmental Surveillance in New Brunswick, 1978-79. Effects of Spray Operations for Forest Protection against Spruce Budworm. Committee for Environmental Monitoring of Forest Insect Control Operations (EMOFICO). University of New Brunswick. p.37-40.
- Plowright, R.C. and L.K. Hartling. 1981. Red clover pollination by bumble bees: a study of the dynamics of a plant pollinator relationship. J. Appl. Ecol. 18: 639-647.
- Plowright, R.C. and B.A. Pendrel. 1978. A comparison of the effects of aminocarb and fenitrothion on forest pollinators in New Brunswick. In V.N. Mallet (ed.). Proceedings of the symposium on aminocarb: Effects of its use on environmental quality. University of Moncton, New Brunswick. 145 p.
- Plowright, R.C., B.A. Pendrel, and I.A. McLaren. 1978a. The impact of aerial fenitrothion spraying upon the population biology of bumble bees (*Bombus* Latr.: Hym.) in southwestern New Brunswick. Can. Entomol. 110: 1145-1156.
- Plowright, R.C., B.A. Pendrel, and G.R. Thaler. 1978b. The impact of spruce budworm control operations on forest pollination in New Brunswick during 1977. Report to the Maritimes Forest Research Centre, Fredericton, N.B. (Contact No. 07SU3KL003-7-0010 Department of Supply and Services, Government of Canada). 82 p.
- Plowright, R.C., D.P. Pielou, I.A. McLaren, and B.A. Pendrel. 1977. The impact of fenitrothion spraying upon the population biology of bumble bees (*Bombus* Latr.: Hym.) in southwest New Brunswick. Report to Dept. Fisheries and the Environment, Government of Canada. 57 p.
- Plowright, R.C. and F.H. Rodd. 1980. The effect of spruce budworm control operations on Hymenoptera in New Brunswick. Can. Entomol. 112: 259-270.
- Plowright, R.C. and G.R. Thaler. 1979. The effects of biocides on forest pollination in New Brunswick. Proceedings of the 4th International Symposium on Pollination, Maryland Agric. Exp. Stn. Spec. Misc. Publ. 1: 483-487.
- Plowright, R.C., J.D. Thomson, and G.R. Thaler. 1980. Pollen removal in *Cypripedium acaule* (Orchidaceae) in relation to aerial fenitrothion spraying in New Brunswick. Can. Entomol. 112: 765-769.

- Poole, C.F., W.J. Carroll, and A.T. Rowe. 1981. Report of the Royal Commission on Forest Protection and Management. Province of Newfoundland. 114 p.
- Proctor, M. and P. Yeo. 1973. The Pollination of Flowers. Collins, London. 418 p.
- Regupathy, A., T.R. Subramaniam, and G. Santharam. 1973. In vitro studies on the effects of certain pesticides on pollen germination and pollen tube growth in brinjal. Madras Agr. J. 60: 634.
- Renval, S. 1977. Insecticides in bees and honey [in Swedish]. Vaxtskyddsnotiser 41: 77-78.
- Romankow, W., F. Wojtowski, and Z. Wilkaniec. 1979. The influence of chemical treatment of harmful and pollinating insects on lucerne seed plantations. Roczn. Nauk Roln. 9: 229-246.
- Schricker, B. 1969. The effect of sublethal doses of parathion on the indication of distance in the honey bee. XXII Int. Beekeeping Congr., Munich. p. 568. Apimondia Publishing House, Bucharest, Rumania.
- Schricker, B. 1972. The effect of sublethal doses of parathion E-605 on the chronological memory of the honey bee. Insect. Soc. 19: 411.
- Schricker, B. 1974a. Der Einfluss subletalen Dosen von Parathion (E 605) auf das Zeitgeächtnis der Honigbiene. [The effect of sublethal doses of parathion (E 605) on the time sense in the honey bee.] Apidologie 5: 385-398.
- Schricker, B. 1974b. Der Einfluss subletalen Dosen von Parathion (E 605) auf die Entfernungsweisung bei der Honigbiene. [The effect of sublethal doses of parathion on the indication of distance in honeybees.] Apidologie 5: 149-175.
- Schricker, B. and W.P. Stephen. 1970. The effects of sublethal doses of parathion on honeybee behaviour. 1. Oral administration and the communication dance. J. Apic. Res. 9: 141-153.
- Sexsmith, W.A. 1987. The evolution of buffer zones for forest insect spraying in New Brunswick. In P.D. Kingsbury and J.G. Trial (eds.). Buffer zones: their application to forest insect control operations, Proceedings of the Buffer Zone Workshop, Eastern Spruce Budworm Council. p. 4-12.
- Skepasts, A.V., D.W. Taylor, and G.T. Bowman. 1987. The alfalfa leafcutting bee, *Megachile rotundata*, and alfalfa seed production in Timiskaming District, Northern Ontario. Proc. Entomol. Soc. Ont. 118: 161-165.
- Skrocki, C. 1972. Control of beetles on winter rape, with special reference to the turnip seed weevil (*Ceuthorrhynchus assimilis* Payk.). [In Polish.] Roczn. Nauk Roln. E (1972) 2: 21-32.
- SOMER 1985. Effets des insecticides fénitrothion et aminocarbe sur la fructification de quatre espèces végétales en milieu forestier. Final Report to Gouvernement du Québec, Ministère de l'Énergie et des Ressources. SOMER, Société Multidisciplinaire d'Études et de Recherches de Montréal Inc., Montréal. xiv + 208 p.
- Stanley, R.G. and H.F. Linskens. 1974. Pollen Biology Biochemistry Management. Springer-Verlag, Berlin. 307 p.
- Stephen, W.P. and B. Schricker. 1970. The effect of sublethal doses of parathion. II. Site of parathion activity, and signal integration. J. Apic. Res. 9: 155-164.
- Stevenson, J.H. 1978. The acute toxicity of unformulated pesticides to worker honey bees (*Apis mellifera*). Plant Pathol 27: 38-40.
- Stevenson, J.H. and J. Walker. 1976. Oil seed rape and honey-bee poisoning. Proceedings of the 8th British Insecticide & Fungicide Conference, Droitwich, U.K. p. 497-501.
- Sundaram, K.M.S. 1974. Determination of fenitrothion residues in honeybees and their products. Chem. Control Res. Inst., Inf. Rep. CC-X-63, Can. For. Serv. 23 p.

- Sundaram, K.M.S. 1975. Fenitrothion residues in honeybees and their products collected from forested areas sprayed with the insecticide for budworm control. Chem. Control Res. Inst. Inf. Rep. CC-X-89, 27 p.
- Takeuchi, K., M. Higo, and T. Sakai. 1981. Susceptibility of honeybees to insecticides. Tamagawa Daigaku Nogakubu Kenkyu Hokoku 20: 40-46 (on request).
- Taylor, K.S., G.D. Waller, and L.A. Crowder. 1987. Impairment of a classical conditioned response of the honey bee *Apis mellifera* L. by sublethal doses of synthetic pyrethroid insecticides. Apidologie 18: 243-252.
- Thakur, A.K., N.P. Kashyap, and C.S. Dogra. 1981. Relative toxicity of some pesticides against Italian bees, *Apis mellifera*. Indian Bee J. 43: 101-103.
- Thaler, G.R. and R.C. Plowright. 1980. The effect of aerial insecticide spraying for spruce budworm control on the fecundity of entomophilous plants in New Brunswick. Can. J. Bot. 50: 2022-2027.
- Thomson, J.D. and R.C. Plowright. 1980. Pollen carry over, nectar rewards, and pollinator behaviour with special reference to *Diervilla lonicera*. Oecologia 46: 68-74.
- Thomson, J.D., R.C. Plowright, and G.R. Thaler. 1985. Matacil insecticide spraying, pollinator mortality, and plant fecundity in New Brunswick forests. Can. J. Bot. 63: 2056-2061.
- Thorpe, E.J. 1979. The effects of forest spraying on solitary bee (Hymenoptera: Apoidea) populations in New Brunswick. M.Sc. thesis, University of New Brunswick, Fredericton, New Brunswick. 94 p.
- Thorpe, E.J. 1980. Effects of spraying on solitary bee populations in forests. In I.W. Varty (ed.). Environmental Surveillance in New Brunswick, 1978-79. Effects of Spray Operations for Forest Protection against Spruce Budworm. Committee for Environmental Monitoring of Forest Insect Control Operations (EMOFICO). University of New Brunswick. p. 41-42.
- Torchio, P.F. 1973. Relative toxicity of insecticides to the honeybee, alkali bee, and alfalfa leafcutting bee. Kans. Entomol. Soc. 46: 446-453.
- Varty, I.W. and N.E. Carter. 1974. Inventory of litter arthropods and airborne insects in fir-spruce stands treated with insecticides. Maritimes For. Res. Centre, Inf. Rep. M-X-48, Can. For. Serv. 32 p.
- Whitehead, D.R. 1969. Wind pollination in the angiosperms: Evolutionary and environmental considerations. Evol. 23: 28-35.
- Whitehead, D.R. 1983. Wind pollination: Some ecological and evolutionary perspectives. In L. Real (ed.). Pollination Biology. Academic Press, Orlando, p. 97-109.
- Winston, M.L. and C.D. Scott. 1984. The value of bee pollination to Canadian apiculture. Can. Beekeeping 11: 134.
- WLR 1985. Negligence: Tutton v. A.D. Walter Ltd: QBD (D.R.M. Henry QC, sitting as deputy High Court Judge): 12 October 1984. The Law Society's Gazette, 4 December 1985. p. 3535.
- Wood, G.W. 1965. Note on the activity of native bee pollinators in relation to the bloom period of lowbush blueberry. J. Econ. Entomol. 58: 777.
- Wood, G.W. 1974. Information supplied to the expert panel on fenitrothion. NRCC Publ. No. NRCC 14104, Ottawa.
- Wood, G.W. 1977. The effects of the fenitrothion spray program, on bees and pollination. Proceedings of a Symposium on Fenitrothion. NRCC Publ. No. NRCC 16073, Ottawa. p. 321-333.
- Wood, G.W. 1979. Recuperation of native bee populations in blueberry fields exposed to drift of fenitrothion from forest spray operations in New Brunswick. J. Econ. Entomol. 72: 36-39.

- Wood, G.W. 1980a. Monitoring of spray drift in cleared areas near forest sprayed with fenitrothion in 1980. Unpublished report submitted to the New Brunswick Dept. Environment. 11 p.
- Wood, G.W. 1980b. Monitoring of insecticide drift from forest spray operations and its effect on lowbush blueberry in New Brunswick in 1980. Unpublished report submitted to the New Brunswick Dept. Environment. 15 p.
- Wood, G.W. 1980c. Pollination Surveys in Blueberry Fields, 1979. In I.W. Varty (ed.). Environmental Surveillance in New Brunswick, 1978-79. Effects of spray operations for forest protection against spruce budworm. Committee for Environmental Monitoring of Forest Insect Control Operations (EMOFICO). University of New Brunswick. p.40-41.
- Wood, G.W. 1980d. Review of Pollinators and Forest Spraying. In I.W. Varty (ed.). Environmental Surveillance in New Brunswick, 1978-79. Effects of spray operations for forest protection against spruce budworm. Committee for Environmental Monitoring of Forest Insect Control Operations (EMOFICO). University of New Brunswick. p. 36-37.
- Wood, G.W. 1980e. Bee toxicity from fenitrothion aerosol. In I.W. Varty (ed.). Environmental Surveillance in New Brunswick, 1978-79. Effects of spray operations for forest protection against spruce budworm. Committee for Environmental Monitoring of Forest Insect Control Operations (EMOFICO). University of New Brunswick. p. 60.
- Wood, G.W. 1982a. A review of forest spraying and its effect on pollinators. In D.R. Townsend (ed.). Environmental Surveillance of Spray Operations for Forest Protection in New Brunswick, 1980-81. Committee for Environmental Monitoring of Forest Insect Control Operations (EMOFICO). Environment New Brunswick. p. 66-68.
- Wood, G.W. 1982b. Monitoring for harmful effects of the 1980 budworm spray program on blueberry production in New Brunswick. In D.R. Townsend (ed.). Environmental Surveillance of Spray Operations for Forest Protection in New Brunswick, 1980-81. Committee for Environmental Monitoring of Forest Insect Control Operations (EMOFICO). Environment New Brunswick. p. 69.
- Wood, G.W., K.B. McRae, and E.N. Estabrooks. 1979. Area assessment of blueberry production in New Brunswick. Can. J. Agric. Econ. 27: 85-91.
- Wood, G.W. and D.K.R. Stewart. 1976. Fenitrothion residues in blueberry fields after aerial forest spraying. Bull. Environ. Contam. Toxicol. 15: 623-629.

CHAPTER THREE

FENITROTHION EFFECTS ON FOREST SONGBIRDS: A CRITICAL NEW LOOK

D. G. Busby¹, L. M. White², P. A. Pearce¹,
and P. Mineau³

- 1 Canadian Wildlife Service
Conservation and Protection
Environment Canada
Fredericton, New Brunswick
- 2 Halifax, Nova Scotia
- 3 Canadian Wildlife Service
National Wildlife Research Centre
Conservation and Protection
Environment Canada
Ottawa, Ontario

TABLE OF CONTENTS

LIST OF TABLES	46	3.8.3 Acute Responses in Wild Birds	90
LIST OF FIGURES	46	3.8.4 Insecticide Residue Analyses	92
3.0 SUMMARY	47	3.9 POPULATION EFFECTS	93
3.1 INTRODUCTION	49	3.10 CONCLUSIONS	95
3.2 AERIAL APPLICATION OF FENITROTHION IN CANADA: A HISTORICAL PERSPECTIVE	49	3.11 ACKNOWLEDGEMENTS	95
3.3 TOXICOLOGY	54	3.12 LITERATURE CITED	95
3.3.1 Mode of Action and Avian Metabolism	54	3.13 APPENDICES	107
3.3.2 Acute and Subacute Responses	56		
3.3.3 Recovery of Brain ChE Activity	61		
3.4 ENVIRONMENTAL RESIDUES AND FATE	61		
3.5 SPRAY DEPOSITION IN THE ENVIRONMENT	63		
3.6 AVIAN IMPACT ASSESSMENT TECHNIQUES	64		
3.6.1 Carcass Searches	64		
3.6.2 Population Censuses	65		
3.6.2.1 Singing-bird surveys	65		
3.6.2.2 Territory mapping	66		
3.6.2.3 Netting studies	66		
3.6.3 Nesting Studies	66		
3.6.4 Observations of Abnormal Behaviour	67		
3.6.5 Experiments with Caged Birds	67		
3.6.6 Brain Cholinesterase Activity Assessments	68		
3.7 ROUTES OF EXPOSURE	69		
3.8 DATA SPECIFIC TO FENITROTHION USE IN FORESTRY	71		
3.8.1 Brain Cholinesterase Data	71		
3.8.1.1 Spray dosage rate	71		
3.8.1.2 Forest structure and topography	73		
3.8.1.3 Influence of spray method	73		
3.8.1.4 Collection bias	77		
3.8.1.5 Number of spray applications	81		
3.8.2 Reproductive and Behavioural Effects	82		
3.8.2.1 Territorial behaviour	83		
3.8.2.2 Reproductive success	85		
3.8.2.2.1 Nesting studies	87		
3.8.2.2.2 Netting studies	88		
3.8.2.2.3 Nestling growth	89		
3.8.2.3 Effects of insecticide induced food depression	89		

LIST OF TABLES

- 3.1 Fenitrothion LD50s for various bird species.
- 3.2 Fenitrothion LC50s for various bird species.
- 3.3 Time for recovery of brain ChE activity to 80% and 100% of control levels in bird species for various OPs.
- 3.4 Proportions of birds having brain ChE depression $\geq 20\%$ and $\geq 50\%$ in relation to dosage rate of fenitrothion.
- 3.5 Proportions of birds having brain ChE depression $\geq 20\%$ and $\geq 50\%$ in relation to dosage rate of fenitrothion, aircraft type, spray equipment and timing of spray applications.
- 3.6 Proportions of White-throated Sparrows with $\geq 20\%$ and $\geq 50\%$ brain ChE depression in relation to dosage rate of fenitrothion, aircraft type and spray equipment.
- 3.7 Proportions of Tennessee Warblers with $\geq 20\%$ and $\geq 50\%$ brain ChE depression in relation to dosage rate of fenitrothion, aircraft type and spray equipment.
- 3.8 Proportions of Magnolia Warblers with $\geq 20\%$ and $\geq 50\%$ brain ChE depression in relation to dosage rate of fenitrothion, aircraft type and spray equipment.
- 3.9 Proportions of Bay-breasted Warblers with $\geq 20\%$ and $\geq 50\%$ brain ChE depression in relation to dosage rate of fenitrothion, aircraft type and spray equipment.
- 3.10 Maximum brain ChE depression in birds in relation to day collected after sprays of fenitrothion.
- 3.11 The bias in impact assessments from inclusion of birds collected before the onset of brain ChE depression (i.e. day 0; area sprayed with 210 g a.i./ha fenitrothion by TBM aircraft fitted with rotary atomizer spray equipment).
- 3.12 Comparison of songbirds having $\geq 20\%$ and $\geq 50\%$ brain ChE inhibition after 1st and 2nd sprays of fenitrothion for spruce budworm and hemlock looper control.
- 3.13 Comparison of White-throated Sparrows having $\geq 20\%$ and $\geq 50\%$ brain ChE inhibition after 1st and 2nd sprays of fenitrothion for spruce budworm and hemlock looper control.
- 3.14 The fate and territoriality of White-throated Sparrows within 2 d postspray (unless otherwise noted) after application of 420 g a.i./ha fenitrothion and a second application 8 d later of 210 g a.i./ha.
- 3.15 Hatching, fledging and overall reproductive success of White-throated Sparrows in control and sprayed (fenitrothion at 420 g a.i./ha + 210 g a.i./ha 8 d later) areas.
- 3.16 Birds found sick, moribund, or dead in spray blocks after fenitrothion spraying of forests in New Brunswick, Newfoundland and Maine, 1967-1987.
- 3.17 Fenitrothion residues (ppm wet weight) in songbirds found sick (S), moribund (M), or dead (D) following fenitrothion sprays in New Brunswick, 1970-1976 (Canadian Wildlife Service, unpublished data).

LIST OF FIGURES

- 3.1 Map of New Brunswick showing the extent of the 1976 spruce budworm spray program (shaded area), the largest spray program in that province to date.
- 3.2 Area of New Brunswick forest flown and sprayed by Forest Protection Limited, 1978-1987.
- 3.3 Proportion of area of New Brunswick forest flown by Forest Protection Limited, 1978-1987, by insecticide.
- 3.4 Relative amounts of fenitrothion and aminocarb (active ingredient) sprayed on New Brunswick forest by Forest Protection Limited, 1978-1987.
- 3.5 Map of New Brunswick showing total amounts of fenitrothion (active ingredient) sprayed on forest for spruce budworm control, 1968-1987.
- 3.6 Theoretical routes of insecticide exposure in songbirds and stages of reproduction potentially affected through ChE inhibition in adults and young.

3.0 SUMMARY

Fenitrothion, the main chemical insecticide used in the last two decades to protect eastern Canadian forests against defoliating insects, has attracted considerable scientific attention. This document attempts to review fenitrothion effects on forest avifauna.

Fenitrothion's primary mode of action, like that of other organophosphorus insecticides (OPs), is cholinesterase (ChE) inhibition. There is little tendency for it to accumulate in avian tissues despite its moderately lipophilic nature. Conventional laboratory toxicity data are generally inadequate in predicting the risk to birds in field situations. Altricial birds are most vulnerable to forest spraying but are virtually unrepresented in standard laboratory test procedures. Experimental protocols in laboratory tests minimize the many environmental stresses that wild birds are subject to. Furthermore, standard toxicity testing focuses almost exclusively on the insecticide's active ingredient. The toxicity of the formulated product, as used in protection operations, receives little attention. Although lethality is clearly an important end point and receives a great deal of interest, recent investigations show the importance of sublethal effects in birds, such as anorexia, altered behavioural responses and compromised physiological ability to deal with natural stressors.

The rates and pathways of fenitrothion degradation in the environment are relatively well understood. Although ephemeral in most forest compartments, fenitrothion may persist for a year or more in conifer foliage and the litter-soil matrix. The implications of low-level residual fenitrothion for forest birds have not been explored. Theoretically, they are unlikely to be of major concern.

The relationship between dosage rate and insecticide deposit in operational spray applications is complex and unpredictable. Sprays emitted above a diverse forest canopy are seldom deposited evenly throughout the forest. Parameters of stand structure, topography, local meteorological conditions, and spray technology may all influence insecticide deposit and, therefore, the exposure of birds. Large variations in deposit occur within and among spray regimes. Knowledge of the theoretical dosage rate is consequently inadequate for the reliable prediction of bird responses to a given spray regime. Although increased control and predictability of spray deposit would facilitate environmental impact studies, great variability in insecticide deposit throughout the target forest is inherent and inevitable in aerial

forest spraying. The forestry insecticide registration process has not addressed that problem.

Most forest spray impact assessment techniques have biases which underestimate the effects on songbirds. Searches for dead or moribund birds may provide valuable insight into spray impacts particularly when they are severe. Finding a small number of dead birds in spray zones is reason to suspect a substantial impact, while failure to find any sick or dead birds does not mean that mortality has not occurred. Avian mortality has been noted at all operational dosage rates used. Analyses of avian carcasses for residual fenitrothion are only minimally useful because of the labile nature of fenitrothion and the high cost of analyses. Problems inherent in census methods render singing-bird surveys capable of detecting only major effects. Territory mapping techniques using marked birds are valuable but labour intensive. Also, any method involving the capture and marking of birds is not without biases, a poor representation of high canopy species being the most serious. The deployment of caged birds in the spray zone as a practical approach to spray monitoring is fraught with logistical and technical difficulties. Methods involving the capture of birds by mist nets are potentially useful, but require much effort and have biases which make interpretation of results difficult. Studies of reproduction, although requiring intensive and sustained effort by the investigator, provide essential information on the process underlying population integrity. Observation of abnormal behaviour in wild birds may provide a limited means of evaluating insecticide impacts. However, the relationship between insecticide toxicity and the onset of behavioural symptoms in birds is complex and poorly documented.

The measurement of brain ChE activity, directly linked to the mode of action of fenitrothion, is the most useful indicator of exposure. It provides a monitoring tool on which a meaningful impact statement can be based. However, the least affected birds are more likely to be sampled and, like most other field monitoring methods, the technique favours underestimation of impact. Sampling protocols must be designed to account for interspecific variability in sensitivity and vulnerability to the insecticide.

At operational dosage rates, acute responses of birds to fenitrothion spraying may include reduced singing and activity, incapacitation, and mortality. With the advent of buffer zones (removing spraying from areas near human habitation), and the switch of investigations to focused research from extensive monitoring, dead birds are less frequently found following fenitrothion spraying than in the past.

Finding dead birds has been more fortuitous, as in recent studies casualty searches have not been conducted.

The limitations of the spray monitoring techniques employed underline the inadvisability of relying on any one method. Determining the possible influence of fenitrothion spraying on bird populations in the long term is difficult and inadequately addressed. To determine the effects of forest spraying at the bird population level, it is necessary to assess reproductive performance in several indicator bird species.

After spraying, birds are exposed to fenitrothion through ingestion, inhalation, and dermal contact. At present, risk scenarios for songbirds cannot be developed because information on the relative importance of those pathways is lacking. Factors include the level of air contamination, the amount and pattern of spray deposit, the birds' foraging and nesting micro-habitats, the birds' maturity, and the birds' activity in relation to the timing of the spray. Until their relative importance can be determined, predictive models cannot be constructed.

Fenitrothion applied at dosage rates of 140-420 g a.i./ha may result in statistically significant depression of brain ChE activity in songbirds. A 20% or greater depression indicates exposure to the insecticide. A 50% or greater depression is indicative of a life-threatening level of exposure although some birds survive higher levels of inhibition and others succumb at levels below 50%. Even if the exposed population could be adequately sampled, field situations make it difficult to predict the magnitude of direct lethality on the basis of ChE measurements.

All fenitrothion spray events monitored demonstrate significant exposure of at least part of the bird population. In all spray blocks monitored, some birds exceeded the 50% depression criterion for potential lethality. The proportion found to have a 20% or more depression in brain ChE activity varied tremendously among spray blocks even when all the components of spray equipment and dosage rate were constant. In contrast, after first and second sprays in a split application within a given block, a similar proportion of birds displayed a 20% or more depression. That difference suggests that characteristics of the spray block itself may be important variables influencing exposure of, and hence impact on, birds.

Following first and second sprays in a split application, the proportion of birds exhibiting brain ChE inhibition of greater than 50% was lower after

the second spray. That difference may be due to selective mortality or severe debilitation of those individuals more vulnerable to the spray as a result of their habits, their nest location, or their utilization of the micro-habitat, and hence their removal from the population available for sampling. If that interpretation is correct, a certain level of undetected loss in the adult breeding population is associated with most fenitrothion applications.

Evidence suggests that the prescribed 5 or 6 d interval between sprays may be insufficient to allow birds to recover. The method of fenitrothion delivery and emission may be important in contributing to the risk to songbirds. The wide variety of operational spray regimes has done much to hamper scientific investigation because it has been difficult to isolate the effect of any single variable.

Songbird reproduction is often synchronized with periods of insect abundance - the time when forest insecticide spraying takes place. Various stages of the breeding cycle, including territory maintenance, incubation, and parental care of young may be affected, resulting in impairment or failure of the breeding attempt. Dosage rates of 300-420 g a.i./ha have been associated with such effects. Insufficient studies have been done on the effects of insecticides on nestling songbird growth and development, a potentially useful area for investigation. Forest songbirds may adjust their foraging locations in response to reduced insect food availability, but the energetic implications for reproduction and survival of birds showing that response have not been investigated.

In White-throated Sparrows, a mean brain ChE depression of 42% was found in birds exposed to the same spray as those exhibiting a broad range of effects including mortality of adults and young, nest desertion, desultory incubation, and ultimately, lowered reproductive success. That observation argues for placing a limit of acceptability on brain ChE activity depression averaging 40-50% in the sampled bird population. Under that criterion, continued broadscale use of fenitrothion in forestry is questionable since that level of brain ChE depression is regularly found in birds sampled in forests sprayed with fenitrothion.

3.1 INTRODUCTION

This document serves two purposes. First, it presents a compilation of a scattered and diverse literature base in order to facilitate a critical assessment of fenitrothion effects on birds, particularly forest songbirds. Second, it presents a re-analysis of some data from the open literature and incorporates data from Canadian Wildlife Service studies not previously published. Thus, it goes beyond a "review" in the strict sense.

The literature base for fenitrothion is extremely broad, extending from its basic chemistry, environmental fate, toxicology, efficacy against insect pests, animal metabolism and physiology, to environmental effects. The sections on toxicology, environmental fate and residues, spray deposition in the environment, and routes of exposure are not exhaustive reviews of the literature. Rather, they give the reader a necessary general background in the fundamental principles necessary to interpret subsequent sections.

This report focuses almost exclusively on the insecticide fenitrothion. Where necessary and informative, literature concerning other OPs and carbamate insecticides is discussed. There is no attempt to provide complete coverage of those insecticides or to assess fenitrothion effects in animal groups other than birds.

3.2 AERIAL APPLICATION OF FENITROTHION IN CANADA: A HISTORICAL PERSPECTIVE

Canada began aerial application of pesticides in forests when, in the late 1920s, Nova Scotia mounted control operations using calcium arsenate against the spruce budworm (*Choristoneura fumiferana*). The mid-1940s brought extensive operational control programs against forest insects when surplus military aircraft and the "miracle insecticide" DDT became widely available (Nigam 1975).

With steadily increasing demands on the forest resource, protection programs increased in number and extent. In recent times, operational or experimental aerial spray applications spanned Canada from Newfoundland to British Columbia's Queen Charlotte Islands (Prebble 1975).

Although several insect pests have been targeted, spruce budworm control programs in eastern Canada have been the largest, resulting in the most continued use of chemical insecticides in forestry. A new generation of insecticides - the less persistent but more acutely toxic OPs and a carbamate - replaced DDT, the most widely used insecticide in the 1950s and 1960s. Among the OPs, the introduction of phosphamidon for use alongside waterways reduced the harmful effects of DDT on aquatic organisms. Because phosphamidon proved to be highly toxic to birds, fenitrothion replaced it, emerging as the most-used insecticide until the mid 1980s when several provinces began moving to the microbial insecticide *Bacillus thuringiensis* (B.t.).

Fenitrothion (O,O-dimethyl O-4-nitro-*m*-tolyl phosphorothioate) (IUPAC) is registered for use against a variety of forest insect pests in Canada. Several different companies manufacture it and supply it to Canadian forest protection agencies under trade names including Accothion, Novathion, and Sumithion.

Only New Brunswick and Newfoundland currently use fenitrothion in large-scale aerial control programs, against spruce budworm and hemlock looper (*Lambdina fiscellaria*), respectively. In New Brunswick, spray operations were conducted every year since 1952 (except 1959) for budworm control. Hemlock looper exhibits more discrete, shorter outbreaks than spruce budworm, and relatively smaller areas are sprayed.

The spruce budworm control program in New Brunswick is both the greatest use of fenitrothion in forest protection and the largest program of its kind in the world. Aerial application of the insecticide takes place during the latter part of May to mid-June, at dosage rates between 140 and 280 g a.i./ha. During the past two decades, an average of between one and two million hectares of New Brunswick forest have been sprayed annually. That program peaked in 1976, when treatment of 3.9 million hectares with four insecticides in 23 spray regimes occurred (Pearce et al. 1979a). Figure 3.1 shows the extent of the 1976 spray program in New Brunswick. Subsequent spray programs in that province have been considerably smaller and less complex.

Figures 3.2-3.4 illustrate several different aspects of forestry insecticide use in New Brunswick over the past decade. Data for those figures represent spraying by Forest Protection Limited, the official New Brunswick spray company. Excluded from the figures are the freehold lands of J. D. Irving Limited, which

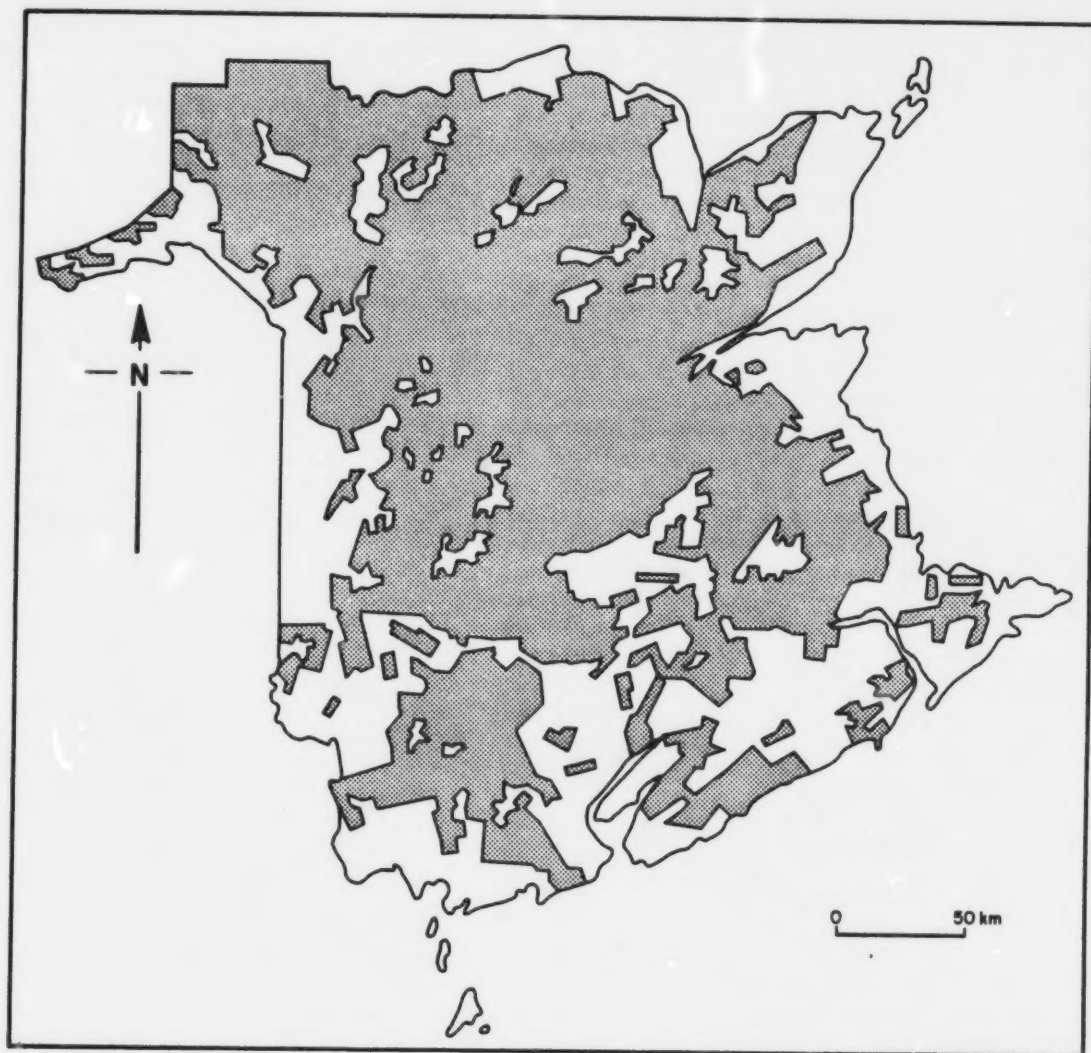


Figure 3.1 Map of New Brunswick showing the extent of the 1976 spruce budworm spray program (shaded area), the largest spray program in that province to date.

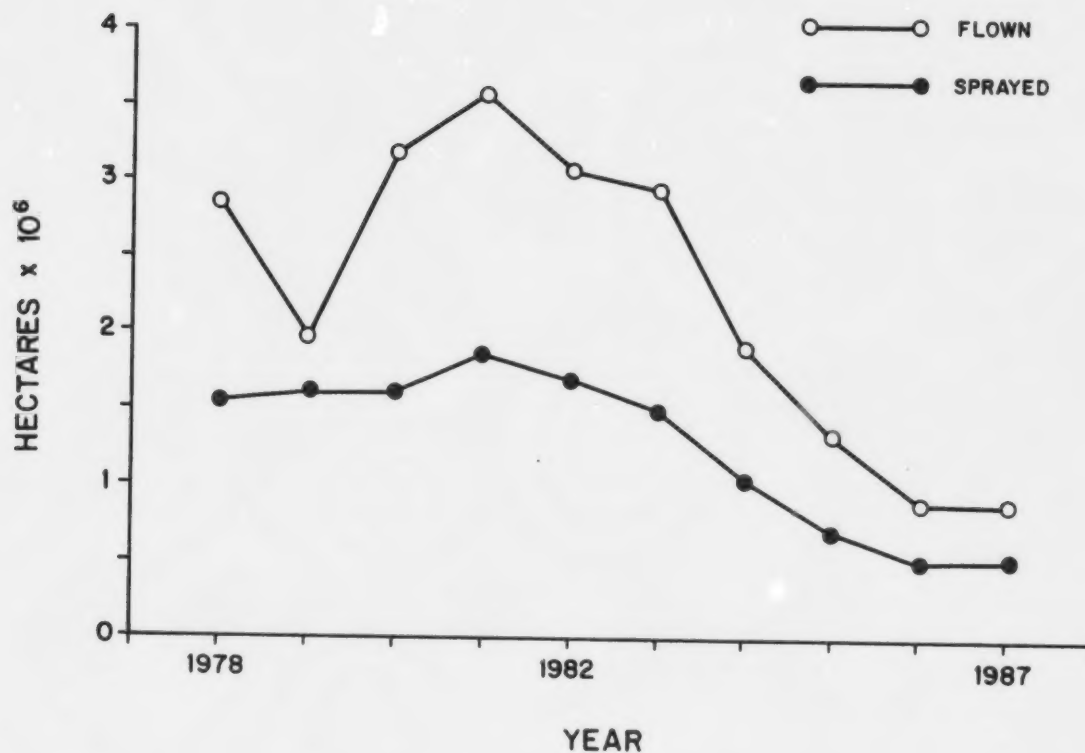


Figure 3.2 Area of New Brunswick forest flown and sprayed by Forest Protection Limited, 1978-1987.

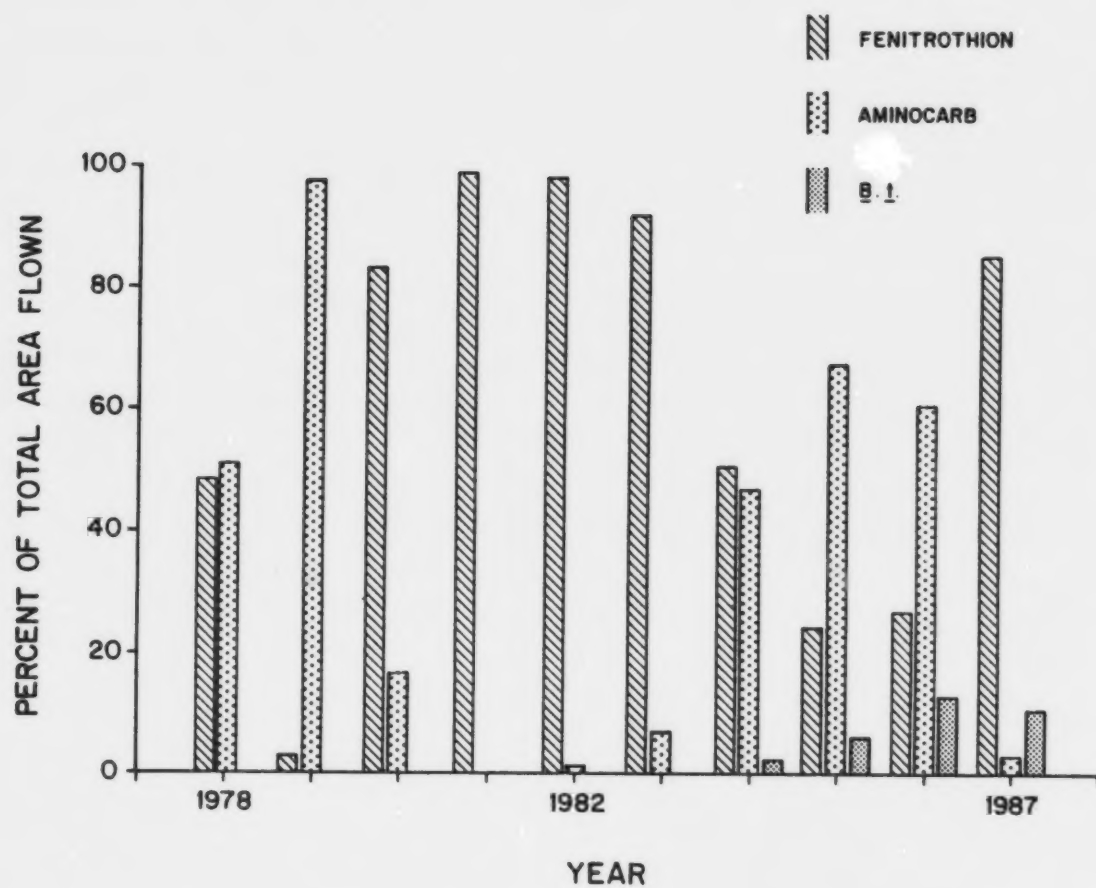


Figure 3.3 Proportion of area of New Brunswick forest flown by Forest Protection Limited, 1978-1987, by insecticide.

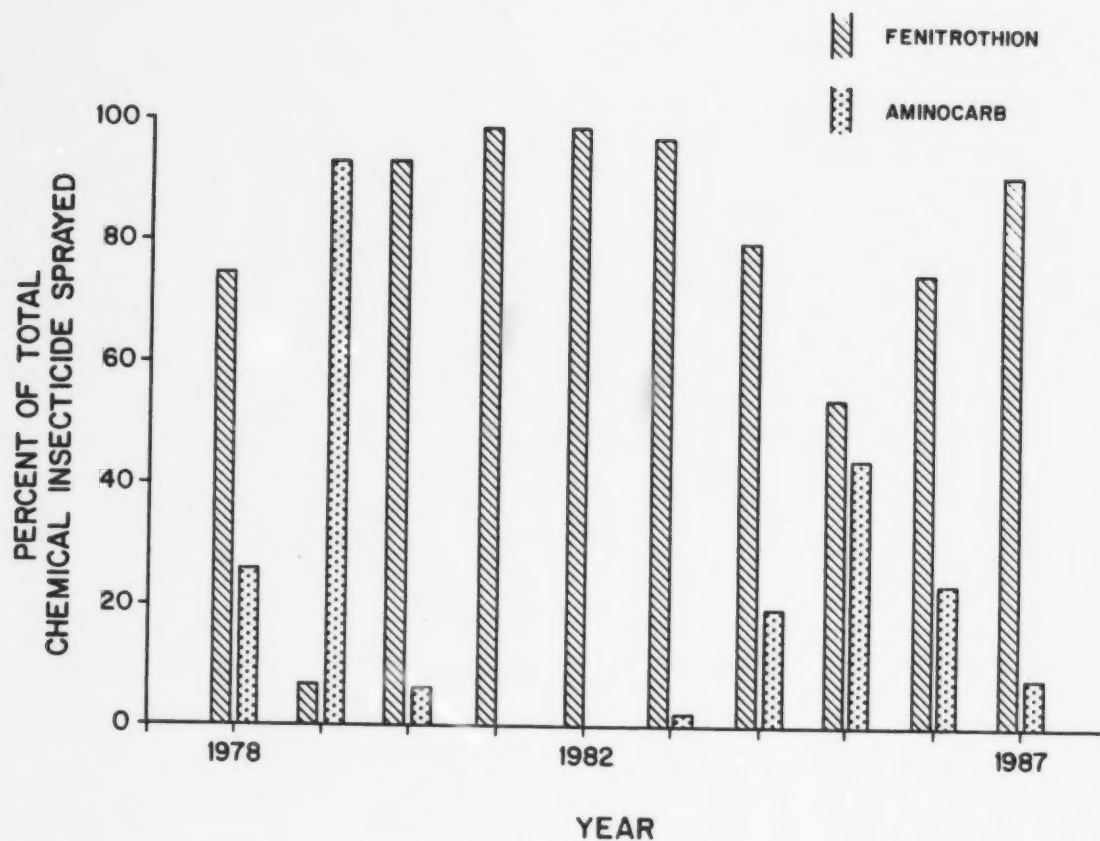


Figure 3.4 Relative amounts of fenitrothion and aminocarb (active ingredient) sprayed on New Brunswick forest by Forest Protection Limited, 1978-1987.

conducts its own aerial control programs against spruce budworm larvae and spruce budmoth (*Zeiraphera canadensis*) larvae and adults. Experimental spraying of insecticides is not addressed by the data presented.

Figure 3.2 illustrates the scale of the New Brunswick spray operations in the past decade. There are two ways of describing the extent of a spray program. "Hectares Sprayed" refers to the total forest area receiving insecticide treatment; and "Hectares Flown" refers to the total area of forest sprayed - a one hectare parcel of land which was treated twice represents two hectares flown. The latter category is more directly related to the amount of insecticide used. In years when some areas are treated only once and when one area is treated twice but with different insecticides, its relative use is easily quantified. In most years, since 1978, the area flown was about twice that of the area sprayed, indicating that most areas were treated twice - the usual spruce budworm control protocol. In 1979, however, the area flown was only slightly greater than the area sprayed, indicating two treatments applied to only a small part of the area.

The three insecticides employed in operational aerial control of spruce budworm in New Brunswick in the past decade were fenitrothion, the carbamate aminocarb, and B.t. (Figure 3.2). Aminocarb virtually replaced fenitrothion in 1979. In 1980, fenitrothion again became the dominant insecticide. Aminocarb replaced fenitrothion in 1984 and dominated for three years. In 1987, fenitrothion use increased sharply.

Changes in insecticide use generally reflect shifts in perceptions concerning both human health and environmental risks and in cost and availability of the insecticides. The first operational spray of B.t. in New Brunswick was in 1984. Although its use has increased since then it had not exceeded approximately 15% of the area flown until 1988, when the area sprayed was evenly split between fenitrothion and B.t.

In terms of actual kilograms of chemical insecticide (active ingredient) sprayed, fenitrothion use exceeded that of aminocarb in all years except 1979 (Figure 3.4), even in those years when the area flown with aminocarb was greater. That reflects a standard dosage rate three times greater for fenitrothion than for aminocarb.

Figure 3.5 shows the total amounts of fenitrothion active ingredient emitted by spray aircraft over New

Brunswick forests during the two decades of that insecticide's use in the province. Included are operational sprays by Forest Protection Limited and by J. D. Irving Limited. Because of fenitrothion's relative non-persistence, discussed below, the map should not be interpreted as indicating insecticide accumulation over time.

Although there is apparently an easing of the current spruce budworm infestation in eastern Canada, it is not inconceivable that future outbreaks may return to the scale of the mid-1970s.

3.3 TOXICOLOGY

3.3.1 Mode of Action and Avian Metabolism

The primary mode of action of OPs, including fenitrothion, is by ChE inhibition. Inhibition results from the phosphorylation of the active sites of the enzyme ChE by the highly toxic metabolite of fenitrothion, fenitrooxon (NRCC 1975). Inhibition of ChE results in an accumulation of acetylcholine (ACh) at nerve synapses causing disruption of nerve function by the continual stimulation of nerve fibres, and the eventual failure of the nerve to repolarize and thus transfer nervous impulses (O'Brien 1960).

Kobayashi et al. (1983) suggested, however, that cholinergic toxicity in birds may, in fact, result from changes in ACh release, metabolism or synthesis, or through modification of the ACh receptor sites rather than through a decrease in ChE activity. Those authors suggested that further studies are required before the biological basis of cholinergic toxicity in birds can be fully understood.

The oxidized metabolite of fenitrothion, fenitrooxon, is formed in both plants and animals. Other metabolites of fenitrothion are present at very low levels as contaminants in the technical formulation but, based on toxicological data derived from animal experiments, are unlikely to influence the toxicity of fenitrothion (NRCC 1975, 1977). The metabolic products of fenitrothion differ according to animal species and environmental conditions (NRCC 1975, 1977).

Birds rapidly metabolize and excrete fenitrothion. Japanese Quail (*Coturnix c. japonica*) were administered ¹⁴C labeled fenitrothion in a single dose of 5 mg/kg body weight, and White Leghorn chickens were given repeated doses of 2 mg/kg for 7 d (Mihara et al. 1979). In both experiments, a 97-

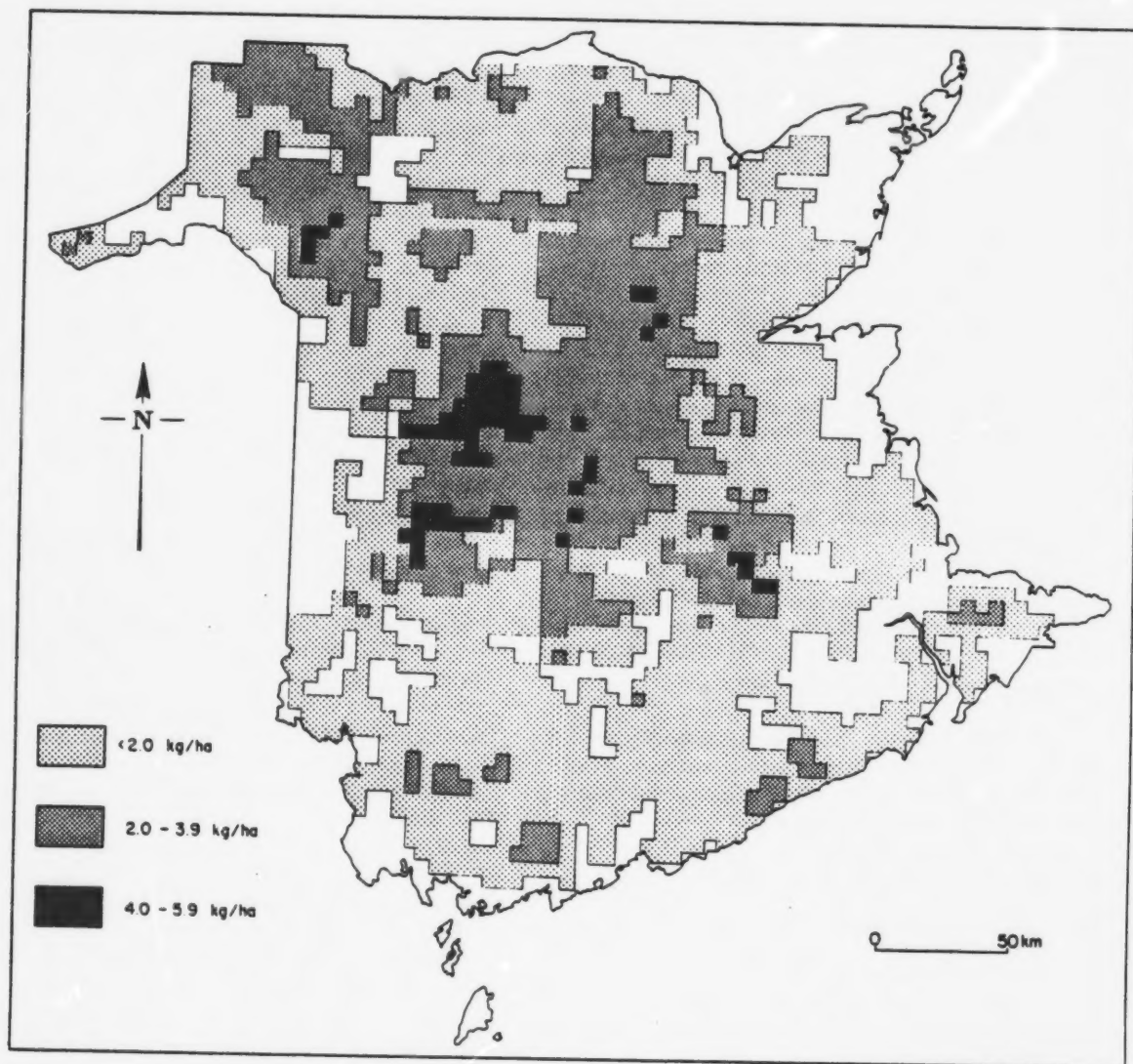


Figure 3.5 Map of New Brunswick showing total amounts of fenitrothion (active ingredient) sprayed on forest for spruce budworm control, 1968-1987.

99% excretion of radioactivity within 1 d was noted and more than 18 metabolites found in the excreta. Serially dosed chickens had only 0.2% of the total dose transferred to the eggs. The gastrointestinal tract of Japanese Quail readily absorbed orally administered ^{14}C labeled fenitrothion which was then rapidly metabolized and showed no tendency for bioaccumulation (Mihara et al. 1979).

Hoffman and Albers (1984) reported that Mallard (*Anas platyrhynchos*) eggs dipped at Day 3 of development in a number of OPs (fenitrothion was not tested) exhibited shortening and contortion of the axial skeleton. It is difficult to determine from the results whether a satisfactory safety margin exists between effect levels and application rates. Kulczycki (1975) reported that an aqueous (0.2%) solution of fenitrothion sprayed on Ring-necked Pheasant (*Phasianus colchicus*) eggs at realistic field rates (unspecified) resulted in significantly reduced hatching success and an increased proportion of problems in hatched chicks. The impact was especially pronounced when the eggs were sprayed late in incubation (e.g. hatching rate of 32% for treatment at 20 d vs. 66% in controls). Hatchlings suffered primarily from paresis (incomplete paralysis) of the legs and paralysis of the neck muscles, symptoms commonly associated with anticholinergic agents in hatched birds. That observation raises the interesting possibility that sufficient quantities of the insecticide can pass through the egg to affect the cholinergic system of both the developed embryo and the hatching chick. Unfortunately, insufficient details were provided to assess the validity of the work or its relevance to the hazard to birds of forest sprays.

3.3.2 Acute and Subacute Responses

Toxicity tests, whether acute or subacute, yield dose-response data condensed to a single number (LD50 or LC50) and determine the level of exposure to a chemical which results in 50% mortality of a test population. Acute tests, commonly referred to as LD50s, involve the administration of a single oral dose designed to measure the inherent toxicity of a chemical. Subacute tests, commonly referred to as LC50s, involve exposure to a diet contaminated with a chemical for a specified period of time. An LC50 test measures the combined effect of a bird's willingness to consume treated food, the inherent toxicity of the chemical, and the duration of exposure (Shirazi et al. 1988). Subacute tests are particularly useful because the toxicant is introduced through the diet, often assumed to be the most significant route of exposure under natural conditions, and

supplemental behavioural observations are also facilitated.

Results of acute and subacute tests often give very different results in terms of relative toxicity, particularly for chemicals which are accumulative in action (e.g. fenitrothion) (Tucker and Crabtree 1970; Heinz et al. 1979; Hudson et al. 1984). Hill et al. (1975) and Heinz et al. (1979) discuss the toxicological relations between acute and subacute results.

Shirazi et al. (1988) provided an extensive discussion of the limitations of LD50 and LC50 tests to provide information on the dose response of organisms to a chemical. Calculation of LD50 and LC50 values involves the transformation of the original dose-response curve omitting potentially useful information outside the mid-range, near low and high dose values. For example, information at low-dose levels indicates whether an organism is metabolizing a chemical quickly or rapidly losing its ability to cope with a chemical metabolically. Thus, those tests are most useful for those organisms whose rate of response to a chemical is maximal in the mid-range of the dose-response curve and which have a flat response near zero.

Tables 3.1 and 3.2 give data on the acute and subacute toxicity of fenitrothion to birds. Mallards apparently are relatively insensitive to fenitrothion and thus may be poor indicators of other species' response to that insecticide. As has been found with many other toxic substances, there is a general inverse relationship between avian weight and sensitivity to fenitrothion, possibly because of higher metabolic rates and food consumption in smaller birds relative to body weight (e.g. Schafer and Brunton 1979). Information on the toxicity of fenitrothion to small passerines, such as warblers, is not currently available, even though those species would be most at risk in field situations. Based on data from toxicity tests of fenitrothion (Tables 3.1 and 3.2), the LD50 for a warbler would be expected to be less than 25 mg/kg body weight and the LC50 would be less than 50 ppm in the diet.

Many factors, such as age, sex, developmental differences, body condition, cold stress, and amount of fat stores can modify an animal's response or sensitivity to a toxic chemical (e.g. Tucker and Haegele 1971; Hudson et al. 1972). Laboratory experiments intentionally minimize the variability associated with such factors by standardizing experimental conditions so that comparisons among different laboratories and different chemicals can be

made (Heinz et al. 1979; Hill and Camardese 1986). Thus, by removing or minimizing the sources of variation in toxic response that would be present

under natural conditions, the hazard of a chemical in natural situations may be underestimated by laboratory studies.

Table 3.1 Fenitrothion LD50s for various bird species.

<i>Species</i>		<i>LD50</i> (mg/kg body wt.)	<i>Source</i> ¹
Mallard	(M) ²	1190	(1)
	(F) ²	1662	(1)
Pheasant	(M)	55.6	(1)
Sharp-tailed Grouse	(M)	53.4	(1)
Bobwhite	(M)	32.0	(1)
	(M)	27.4	(1)
	(F)	23.6	(1)
Redwing Blackbird		25.0	(2)

¹ (1) Hudson et al. 1984; (2) Schafer 1972.

² M = male; F = female.

Table 3.2 Fenitrothion LC50s for various bird species.

<i>Species</i>	<i>LC50</i> (ppm in food)	<i>Source</i> ¹
Mallard	2482	(1)
Pheasant	453	(1)
Japanese Quail	440	(1)
Bobwhite	157	(1)
Common Grackle	78	(2)
White-throated Sparrow	50	(3)

¹ (1) Hill et al. 1975; (2) Grue 1982; (3) D. Forsyth, unpublished data.

Subacute toxicity varies widely among different formulations of the same chemical (Hill and Camardese 1986). Until the actual formulation used in the field has been tested, any conclusions about the lethal hazard associated with field application of a given chemical should be made with reservation (Hill and Camardese 1986). At present, no information is available on the toxicity to birds of fenitrothion formulations used in the field - a matter requiring attention for some time (NRCC 1977). Researchers tend to focus on the active ingredient and fail both to acknowledge the complex nature of the varied formulations and spray regimes used under operational conditions and to examine their toxic properties (NRCC 1977). The use of non-operational formulations may provide unrealistic data for evaluating the impact of field applications.

It is commonly recognized that direct mortality is not always a sensitive measure of a chemical's hazard, as lower levels of exposure may impair reproduction or the ability of an animal to survive in the long term (e.g. Hudson et al. 1984; Galindo et al. 1985). It is difficult to extrapolate data from the LD50 to the hazard of the chemical from use in the field because many factors must be considered. They include dosage rate; contamination of air, water, and vegetation, which may be influenced by the carrier in the pesticide formulation; evenness of application; wind; temperature; vegetation composition; other species present; routes of exposure; persistence in the environment; formation of biologically active metabolites; and degree of accumulation in organisms and the environment (Hudson et al. 1984).

The signs of acute systemic poisoning by OPs are relatively predictable. Acetylcholine is the chemical transmitter of nerve impulses for postganglionic parasympathetic nerve fibres, somatic motor nerves to skeletal muscle, preganglionic fibres of both parasympathetic and sympathetic nerves, and certain synapses in the central nervous system (Murphy 1975). In the laboratory, birds exposed to fenitrothion display the following acute symptoms: ataxia (lack of coordination), high carriage, wing drop, wing shivers, falling, tremors, salivation, loss of righting reflex, tetanic seizures (extreme stiffening of the body), dyspnea (laboured breathing), miosis (contraction of the pupil of the eye), lachrymation (secretion of tears), and wing beat convulsions (Tucker and Crabtree 1970; Hudson et al. 1984).

It is generally thought that ChE inhibition from acute exposure to OPs causes death in air-breathing vertebrates by blocking neurotransmissions in the respiratory centre of the brain or at myoneural junctions of respiratory muscles (O'Brien 1967).

However, mortality in Common Grackles (*Quiscalus quisculus*) from dietary exposure to fenitrothion and other OP chemicals is dose-related and largely the result of pesticide-induced anorexia (Grue 1982). Birds of other species exposed to dietary concentrations of OPs exhibited anorexia (Keith and Mulla 1966; Mehrotra et al. 1967; Hill 1971; Pope and Ward 1972). Starvation was thought to play a major role in the mortality of Red-billed Queleas (*Quelea quelea*) when sprayed with the OP fenitrothion for avicidal purposes (Pope and Ward 1972).

Both passerines (Grue 1982 and references therein) and non-passerines (e.g. Hill 1971; Stromborg 1981 and references therein) frequently experience weight losses of 22-40% when exposed to OPs. The severity and duration of weight loss induced by OP exposure varies among chemicals, concentrations, routes of exposure, species, ages, and between sexes. The above references suggest that anorexia and physiological condition have been important but often overlooked factors in the occurrence of avian mortality resulting from OP exposure in the wild, especially in the case of small passerines.

Dietary factors alter the toxicity of pesticides (Charbonneau and Munro 1983). A recent study by Dahlgren (1988) indicates that the susceptibility of subadult birds to OP poisoning can be influenced by the composition of their diet as juveniles. Feeding on isofenphos-treated seeds severely affected mortality and signs of acute poisoning of subadult Gray Partridges (*Perdix perdix*) fed an insect-poor diet as juveniles. Birds fed an insect-rich diet as juveniles showed no signs of poisoning or mortality. Dahlgren suggests that the differential response may be due to a nitrogen deficiency in the birds on the insect-poor diet as juveniles. Non-specific binding to tissue proteins is an important mechanism used by birds and mammals to detoxify OP compounds (Lauwerys and Murphy 1969a, 1969b). The implications of Dahlgren's findings to both laboratory and field studies need to be explored.

Grue (1982) found seasonal variation in the response of Common Grackles to dietary concentrations of OPs which he attributed to seasonal variation in body-fat reserves. In late summer, birds with low fat reserves, associated with completion of post-nuptial molt and increased ambient temperatures, were more vulnerable to mortality from OP exposure than birds in May which had higher fat reserves. Physiological condition of exposed birds and ambient temperature may influence the toxicity of OP chemicals. Rattner (1982) found that extended periods of limited food availability and exposure to elevated temperatures (36° for 1 d) caused reductions

of 10-17% brain ChE in adult Japanese Quail. Also, OPs apparently alter the ability of organisms to tolerate cold stress, resulting in greater brain ChE depression at lower temperatures (Rattner et al. 1982, 1987; Fleming et al. 1985; Maguire and Williams 1987a, 1987b). Thus, it appears that ambient temperature can potentiate dose-related brain ChE depression, which may result in increased mortality from OP exposure in the wild.

Altricial nestling European Starlings (*Sturnus vulgaris*) appear to be nearly twice as sensitive to OPs as adults (Grue and Shipley 1984), perhaps due to their lower brain ChE levels per unit brain tissue (Grue et al. 1981; Grue and Shipley 1984). However, other factors, such as a poorly developed blood-brain barrier (Bolton 1976) or an inadequate detoxifying system, may be involved in young birds' greater sensitivity to OPs because young precocial birds are more sensitive than adults even though they have similar brain ChE activities (Grue et al. 1981; Grue and Hunter 1984).

In summary, altricial nestlings, which are more sensitive to OPs than older birds, have a more limited thermoregulatory ability and a higher rate of metabolism than their parents and may be particularly vulnerable to otherwise "sublethal" amounts of ChE-inhibiting insecticides.

Because all OPs inhibit ChE activity and thus share a common mechanism of acute toxic action, the known effects of ChE inhibition by different OP chemicals become relevant to assessment of the impact of any particular one. The most frequently used diagnostic measure of the impact of OPs on birds is the brain ChE activity. The ability of OPs to inhibit brain ChE varies among chemicals (Fleming 1981; Fleming and Bradbury 1981), species (Fleming and Grue 1981), age (Grue and Shipley 1984); level of exposure (Fleming and Grue 1981; Busby et al. 1983a); and dietary factors (Dahlgren 1988). Brain ChE activity is similar between the sexes in several species (Ludke et al. 1975; Grue et al. 1982a; Mitchell and White 1982).

Organophosphorus insecticide-induced mortality appears to be dose related, but the degree of brain ChE depression is less clearly related to dose. There apparently is a high degree of variability among individuals of the same species given the same dose although mean brain ChE levels of birds sampled appear to be dose related (Fleming 1981; Fleming and Bradbury 1981; Fleming and Grue 1981; Grue 1982). Variability in response would be expected to be greater where exposure is by dietary intake (e.g.

Grue 1982) because individual food consumption may affect exposure.

The degree of brain ChE inhibition considered lethal varies in relation to such factors as age, physiological condition, and route of exposure. There is agreement that the criteria proposed by Ludke et al. (1975) are sufficient for diagnostic purposes. However, recent discussions in the literature caution against an over-reliance on those fixed criteria (e.g. see Fairbrother and Bennett 1988; Hill 1988a). Ludke and colleagues suggested that brain ChE inhibition of 50% or greater in a dead individual compared to controls is indicative of OP-induced mortality. Because some birds with greater than 50% ChE depression may survive (e.g. Hill and Fleming 1982), brain ChE depression greater than 50% in birds collected while still alive is considered potentially life threatening (Zinkl et al. 1980). Laboratory studies indicate that depression of brain ChE activity greater than 20% (approximately 2 SD below the average value of controls) is indicative of OP exposure in an individual bird (Ludke et al. 1975).

In practice, most laboratory and recorded field mortality is associated with brain ChE inhibition of greater than 50% (e.g. Bunyan et al. 1968; Ludke et al. 1975; White et al. 1979; Grue 1982; DeWeese et al. 1983; Zinkl et al. 1980). Hill (1988b) suggested that the criteria of Ludke et al. (1975) are conservative. However, the following data indicate that fenitrothion-induced mortality of forest songbirds can be associated with brain ChE inhibition of considerably less than 70%; therefore, the criteria of Ludke et al. (1975) are not unduly conservative. Three moribund birds (legs collapsed, breast on ground, tail spread and upright) and one dead bird were found in New Brunswick following aerial application of 420 g a.i./ha fenitrothion in 1979 (Canadian Wildlife Service, unpublished data). An American Redstart (*Setophaga ruticilla*), Canada Warbler (*Wilsonia canadensis*), and Swainson's Thrush (*Catharus ustulatus*) found within 24 h of spray application showed depressed brain ChE activity from 41-61%. In the same study, a 52% depression in brain ChE activity of a White-throated Sparrow (*Zonotrichia albicollis*) found dead 7 d post-spray was measured. Two dead birds, a Pine Siskin (*Carduelis pinus*) and a Yellow-rumped Warbler (*Dendroica coronata*), found in 1987 following aerial application of 210 g a.i./ha fenitrothion in New Brunswick, had brain ChE activity depressed by 76 and 78%. Forsyth (Appendix 3.1) conducted an LC50 test on captured White-throated Sparrows. A diet containing 10, 25, 40, 60, 90, and 100 ppm fenitrothion was fed to groups of ten birds. Mortality was dose

related, and brain ChE activity was depressed 57-81% in birds that died. Survivors, sacrificed at the conclusion of the experiment, had brain ChE activity levels depressed by 3.8-58%.

Holmes (1988) found mortality in caged Zebra Finches (*Poephila guttata*) to be associated with mean brain ChE activity inhibitions of 45.3% (range:

3.3-83.3) and 62.6% (range: 31.9- 83.7), depending on fenitrothion dosage rate.

In three studies of brain ChE depression in forest songbirds in relation to fenitrothion (Busby et al. 1981, 1987a) and aminocarb (Busby et al. 1983b) spraying, statistically significant reductions in brain ChE activity were found even though the mean level

Table 3.3 Time for recovery of brain ChE activity to 80% and 100% of control levels in bird species for various OPs.

Species	Chemical	Dose (mg/kg)	Maximum mean % inhibition of survivors	Recovery time (days)		Source ¹
				80% Control	100% Control	
Mallard	acephate	350	48	5	24	1
	dicrotophos	2.4	57	8	18	1
	fensulfothion	0.75	51	8	27	1
	fonofos	14	33	1	12	1
	malathion	650	33	4	26	1
	parathion	1.75	25	4	33	1
Mallard	dicrotophos	2.4	78	12	32	2
		16 ppm ²	75	11	26	2
	fenthion	5.5	57	10	34	2
		17 ppm ²	66	12	37	2
Mallard	dicrotophos	0.75	55	11	54	3
Barn Owl	dicrotophos	0.3	90	17	67	3
Bobwhite Quail	dicrotophos	3.0	60	15	69	3
Starling	dicrotophos	3.0	64	20	98	3
Common Grackle	dicrotophos	1.0	75	26	109	3
Dark-eyed Junco	acephate	26.52	44	3-6	—	4
Zebra Finch	fenitrothion	1.04	49.5	4.0	50.3	5
		3.8	69.9	6.5	33.5	5

¹ (1) Fleming and Bradbury 1981; (2) Fleming 1981; (3) Fleming and Grue 1981; (4) Zinkl et al. 1981; (5) Holmes 1988.

² Fed to birds in diet for 2 weeks.

of depression was less than 20%, and as little as 12%. Thus, results from field studies do not always fit neatly within the criterion of Ludke et al. (1975) that a brain ChE depression of greater than 20% must be achieved before exposure to an anticholinergic agent is demonstrated.

3.3.3 Recovery of Brain ChE Activity

If an animal survives OP poisoning, ChE activity returns to normal (O'Brien 1960). In birds, most OPs bind irreversibly to the ChE molecule and recovery of ChE activity following exposure depends primarily on *de novo* synthesis of ChE, not as the result of dephosphorylation as previously proposed (Fleming 1981).

The rate of recovery of brain ChE activity depends on the inhibiting chemical and maximum degree of depression (Fleming and Bradbury 1981; Fleming and Grue 1981) but apparently is independent of the duration of exposure (Fleming 1981). Previous exposure to OPs does not appear to significantly affect either the degree of inhibition or recovery of brain ChE activity at subsequent exposures (Fleming 1981). However, there is no information concerning avian response to subsequent exposure before brain ChE activity returned to normal. In eastern Canadian forests, fenitrothion is usually applied in two treatments 5-6 d apart, which may not be sufficient for birds to recover from initial exposure.

In avian species, the general pattern of restoration of brain ChE activity following OP exposure is initial rapid recovery to about 50% of control values followed by much slower recovery to normal levels (Fleming 1981; Fleming and Bradbury 1981; Fleming and Grue 1981). The time taken to recover from exposure was directly related to the initial ChE depression (Fleming and Bradbury 1981; Fleming and Grue 1981). In laboratory studies, the time estimated for 100% recovery of brain ChE activity in birds following exposure to a variety of OPs ranges from 12 to 109 d and for 80% recovery ranges from 1 to 26 d (Table 3.3). Recovery to 80% of normal brain ChE activity is logarithmic when calculated from sample means (Fleming and Bradbury 1981) and follows the model $T = a + b (\log_{10} x)$. However, Fleming (1981) found the fit for the logarithmic curve to be poor for individuals because of the high degree of individual variability in sensitivity to any particular chemical. Further studies are necessary to obtain additional data on the relation between rate of recovery and body size (Fleming and Grue 1981).

Zinkl et al. (1981) noted that Dark-eyed Juncos (*Junco hyemalis*) recovered more rapidly (3 d) from

a single dose of acephate under laboratory conditions than the prolonged (3-33 d) depression of brain ChE activity of songbirds observed in field studies (Zinkl et al. 1977, 1979, 1980). Following aerial applications of fenitrothion at 300 g a.i./ha to lodgepole pine plantations in Scotland, Hamilton et al. (1981) documented significant brain ChE inhibition in some individuals of two species of birds (Chaffinch, *Fringilla coelebs*, and Coal Tit, *Parus ater*) collected as late as 21 d post-spray. That was the last sampling time, and the full duration of the effect could not be determined. For both species, maximum inhibition, in terms of both magnitude and frequency, occurred in the first 2 d following spraying.

The recovery of brain ChE activity of young birds following OP exposure may differ between altricial and precocial species. If they survive OP exposure, young passerine nestlings may recover more rapidly from ChE inhibition than adult passerines because of synthesis of new ChE within brain tissue during development (Grue et al. 1981; Grue and Hunter 1984). The young of precocial species are expected to show the same rate of recovery as adults because of similar brain ChE activity levels. Further studies are necessary to obtain additional information on species-specific attributes affecting the rate of recovery from ChE inhibition.

3.4 ENVIRONMENTAL RESIDUES AND FATE

Although the chemistry of fenitrothion and its metabolites is relatively well understood, clarification of actual rates and pathways of degradation (volatilization, hydrolysis, photolysis, and microbial degradation) is recent. Summarized briefly below is current information regarding the fate of fenitrothion after spray applications.

Prolonged persistence of fenitrothion in natural waters has not been observed (Sundaram 1974a, 1974b; Sundaram and Nott 1984; Holmes et al. 1984) although there are reports of short-term persistence, for about a week, in some aquatic flora and fauna (Kingsbury 1976; Moody et al. 1978; Morrison and Wells 1981; Eidt et al. 1984). Fenitrothion disappears rapidly in stream water through dilution (Eidt and Sundaram 1975), hydrolysis (Sundaram 1973), volatilization (Maguire and Hale 1980), photolysis (Lockhart et al. 1973), and particularly by accumulation in sediments and in aquatic plants and animals (Eidt et al. 1984).

Fenitrothion was undetectable in forest soils 32-64 d after application (Sundaram 1974a; Symons 1977). Degradation products in soils appear to be dependent on both the size and type of microbial populations present as well as the physical characteristics of the soil (NRCC 1977). Rates of disappearance of amino fenitrothion and S-methyl fenitrothion from soils are much more rapid than those of fenitrothion; however, the final degradation products are unknown.

In most forest substrates, fenitrothion does not persist for prolonged periods (Yule and Duffy 1972; Eidt and Sundaram 1975; Miyamoto 1977; Kingsbury 1978; Morrison and Wells 1981; Ayer et al. 1984; Sundaram 1984a, 1987). In contrast, many studies report the relatively long-term persistence and accumulation of fenitrothion in conifer foliage and the litter-soil matrix (Yule and Duffy 1972; Yule 1974; Sundaram 1974a, 1974b, 1984a; Yule and Varty 1975; McNeil et al. 1979; Mallet and Volpé 1982; Sundaram and Sundaram 1982; Morin et al. 1983, 1984; Ayer et al. 1984; Eidt and Mallet 1986; Sundaram and Nott 1986).

Conifer foliage retains fenitrothion in its original form, not as a derivative or metabolite (Prasad and Moody 1976). No breakdown products have been detected in conifer foliage with the exception of trace amounts of fenitrooxon (Yule and Duffy 1972; Sundaram 1984a). Persistence of fenitrothion in foliage and litter is thought to be related to its lipophilic nature, its environmental stability, and its relatively low vapour pressure (Sundaram 1987).

During spray applications, conifer foliage is the primary receptor of fenitrothion. Residues of fenitrothion on spruce and fir foliage decrease by 70-85% in 14 d (Yule and Duffy 1972) and 95% in 10 d (Sundaram 1974a) following the spray. Initially, loss of fenitrothion on conifer foliage is relatively rapid, primarily through volatilization. Subsequent dissipation of fenitrothion occurs through volatilization, the weathering action of humidity, by photolysis and hydrolysis, and, to a lesser extent, by enzymatic degradation (Sundaram 1974a; Sundaram and Sundaram 1982; Sundaram 1984a).

Translocation of fenitrothion within conifer trees apparently does not occur (Prasad and Moody 1976) except into the young foliage of fir, and to a lesser extent spruce, in the year of treatment (Moody et al. 1977). Fenitrothion does not appear to be translocated into developing shoots in the year following treatment (Eidt and Mallet 1986).

Persistent fenitrothion residues reside primarily in the xylem cells of balsam fir, white spruce, and jack pine (Prasad and Moody 1976; Moody et al. 1976). The residues appear to be transferred to the xylem much faster in fir than in other species of conifers (Moody 1978). Subsurface residues are more resistant to such processes as volatilization, photodegradation, and leaching by rain than are surface residues (Sundaram 1986) and can persist at low concentrations from one year to the next (Sundaram 1987).

When optimizing spray deposition of insecticides, it is important not only to increase the total amount deposited on plants or target organisms but also to optimize the ratio of surface (dislodgeable) to subsurface (penetrated) residues (Westlake et al. 1973). The advantage of a high ratio of surface to penetrated residues is that less of the deposit is lost to the environment and more remains for insecticidal activity. The rate and extent of penetration of OP and carbamate insecticides in conifer foliage affects both their insecticidal activity (Robertson and Boetler 1979) and their persistence characteristics (Sundaram and Sundaram 1987).

The amount of surface residues on conifer foliage relates to the dosage rate applied. Surface residues of aminocarb (applied at 90 g a.i./ha) were lower than surface residues of fenitrothion (applied at 210 g a.i./ha) (Sundaram 1986). In contrast, concentrations of penetrated residues were very similar for both insecticides suggesting a maximum level of solubility of insecticides in subsurface layers of conifer foliage (Sundaram 1986; Sundaram and Sundaram 1987). Sundaram and Sundaram (1987) attribute that finding to similarity in the lipophilic nature of fenitrothion and aminocarb, and the solubility of those insecticides in the lipid component of conifer foliage. Thus, there appears to be a maximum limit for subsurface residues which is not affected by an increase in the dosage rates (Eidt and Mallet 1986; Sundaram 1986; Sundaram and Sundaram 1987).

The toxicological significance of low concentrations of fenitrothion persisting in conifer foliage has not been fully explored. McNeil et al. (1979) and Eidt and Mallet (1986) report that such concentrations are toxic to conifer sawflies and become increasingly toxic to the larvae of Swaine jack pine sawfly (*Neodiprion swainei*) with each year's accumulated residues (McNeil et al. 1979). It appears that low-concentration residues are unlikely to affect conifer foliage-eating birds (Eidt and Pearce 1986).

Foliar leaching and leaf fall may be the primary route by which fenitrothion becomes incorporated into the litter-soil matrix (Sundaram 1984a). Fenitrothion was persistent in the litter-soil matrix for up to 50 d after spraying (Yule and Duffy 1972). In Ontario (Sundaram 1984a) and Quebec (Morin et al. 1983), residues of fenitrothion detected in litter one year after spraying showed no evidence of accumulation. Sundaram (1984a) postulated that fenitrothion residues tend to be greater in litter than in soil because of the absorption and dissolution of fenitrothion within the lipoidal humus component of conifer litter. In addition, the greater acidity and hence lower microbial content of conifer litter may contribute to the persistence of fenitrothion in that substrate.

3.5 SPRAY DEPOSITION IN THE ENVIRONMENT

In normal operational spraying, fenitrothion is mixed with solvents, emulsifiers, and diluents. A formulation is the mixture of the insecticide, the solvent used to dissolve it, the emulsifier used to keep the insecticide and solvent in suspension, and the diluent (normally water or oil) used to put the insecticide, solvent, and emulsifier in a form which can be sprayed (NRCC 1977). A spray regime refers to the sequence of treatments (timing, rates, and formulations) applied at prescribed dosages of insecticide by specific aircraft and spray equipment (Pearce and Peakall 1977). Formulation ingredients play a significant role in droplet size spectra, deposit patterns, foliar residues, and persistence characteristics of fenitrothion sprays (Sundaram and Sundaram 1981, 1982; Sundaram and Szeto 1984; Sundaram et al. 1985a, 1985b; Sundaram and Nott 1985; Sundaram and Retnakaran 1987).

During the past decade, there has been an increasing trend towards improving the performance of pesticides by optimizing application methodology through the introduction of ultra low volume (ULV) sprays, i.e. sprays with total volume emitted below 5 L/ha, (Quantick 1985); and ultra ultra low volume (UULV) sprays, i.e. sprays with application rates below 1.47 L/ha, (E. Kettela pers. comm.). Application methodology also changed in response to ever-increasing costs and concerns over minimizing chemical pesticide dispersal in the environment. To maximize efficiency of the insecticide application requires that droplet size spectra, number of drops per unit area, and spray volume deposited on target sites be optimized (Himel and Moore 1967, 1969;

Ware et al. 1970, 1972; Yates and Akesson 1973; Joyce et al. 1977). The relationship between dosage rates and deposition patterns of forest insecticides is very complex and unpredictable. Optimizing formulation properties through the use of various adjuvants is one of the more promising methods to improve the efficiency and predictability of forest insecticide applications.

Deposition may be determined directly using glass plates distributed in the spray area. Gas chromatography is used to analyze acetone or benzene washings from the plates (Wood and Stewart 1976). Alternatively, deposition has been commonly estimated by counting spray droplets on Kromekote cards placed in spray areas and by calculating the amount of fenitrothion from the volume of droplets/unit area, known spray concentrations and measured spread factors (Chansler and Dimond 1971). However, fir and spruce trees are excellent collectors of small spray droplets and tend to be more efficient than the sampling units commonly used (Armstrong 1977). Thus, glass plates or Kromekote cards may be an adequate measure or index of deposition on the ground but may underestimate deposition within the canopy.

If droplets are too large, they give poor coverage, they do not penetrate the micro-habitat of the target organism (Joyce et al. 1977; Barry 1984), and they sediment so rapidly that most of the pesticide falls on the ground instead of on the canopy (Joyce and Beaumont 1978; Matthews 1979; Nguyen and Symons 1984). When drops are too small, there is a greater tendency for them to evaporate before reaching the canopy (Mason 1979). Small drops tend to linger near the canopy for a long time because they have little energy to impact, particularly under still-air conditions (Sundaram and Retnakaran 1987). Consequently, a fine spray would be more susceptible to drift. Not surprisingly, turbulence is a critical factor in the transport and deposition of fine drops in ULV spray applications (Joyce and Beaumont 1978; Spillman and Joyce 1978).

Yule and Cole (1969) reported drift of DDT for several miles during forest spray operations, and Yule et al. (1971) measured low-level fenitrothion contamination of the atmosphere in New Brunswick after a spray. Although fenitrothion residues in balsam fir decreased with distance from the spray block, they were detected up to 14.5 km from the spray area indicating long-distance drift (Morin et al. 1983). Monenco (1978) measured significant residues in water as far away as 30 km from spray areas. Pearce et al. (1979b) measured deposition of

fenitrothion 85 km from the nearest spray zone. They suggested that atmospheric transport and subsequent surface deposition, or the scavenging effect of falling rain, or both, were mechanisms in the redistribution of a spray cloud far beyond the target area.

Aerial forest sprays are emitted above a diverse crown canopy and are seldom deposited evenly throughout the forest. Fudge et al. (1988) found a high degree of variability in fenitrothion deposit using coniferous foliage as a sampling substrate. Large variations in spray deposits can occur from one spray event to another, particularly when volume release rates differ markedly (Himel and Moore 1969; Barry et al. 1977; Spillman and Joyce 1978; Joyce and Beaumont 1978). Many of the sources of variability in spray deposit have been recognized since the earliest experimental spraying of DDT (Haliburton et al. 1975). The relationship between dosage rate and the quantity of insecticide deposited on the forest is extremely complex. It is dependent on such factors as formulation ingredients, application rates, atomizer type, aircraft speed, swath width, accuracy of location of flight paths, macro- and micro-structure of the forest (species composition and the size and density of trees), topography and micro-meteorology above and within the canopy (air temperature, speed, direction, and stability). Armstrong (1977) reviewed the dynamics of spray deposition in relation to many of those factors.

Variability in the actual deposition of fenitrothion on the canopy makes it difficult to detect differences in foliage residues among different spray regimes (Ayer et al. 1984). The location of trees sampled within a forest and the position of the branches sampled are important factors in estimating fenitrothion residues after a spray event. Following aerial applications of insecticides, significantly more spray deposits appear on the upper canopy, on the upwind side of fir trees, and in areas not filtered by other foliage (Hurtig et al. 1953; Armstrong and Yule 1978; Spillman and Joyce 1978; Barry 1984; Sundaram and Nott 1984; Sundaram et al. 1985b; Millikin 1987). Thus, spray deposit is both vertically and horizontally stratified in conifers such that maximum deposition occurs on the upper canopy and foliage on outer branches.

Theoretical dosage rate alone therefore falls far short of conveying enough information to allow consistent, correct prediction of the effects of field operations because of the large variability in deposits reaching the forest. An increase in the control and predictability of spray deposit is necessary before either the environmental effects of insecticide spraying can be fully assessed or the risk/benefit criteria can be realistically determined.

3.6 AVIAN IMPACT ASSESSMENT TECHNIQUES

Grue and Shipley (1981), Peakall and Bart (1983), and Mineau and Peakall (1987) recently reviewed some of the main techniques used to assess the impact of aerial application of forest insecticides on birds, including: carcass searches, population censuses, nesting studies, observations of abnormal behaviour, experiments with caged birds, and brain ChE determinations.

3.6.1 Carcass Searches

In the past, searching for dead or moribund forest birds was a commonly used method to estimate mortality in pesticide-treated areas (Fowle 1965; Pearce 1967, 1968; Buckner 1974). Estimating the magnitude of the kill and confirming the cause of mortality is generally considered essential in determining the environmental hazard of pesticides. But Heinz et al. (1979) stressed that that technique is a necessary but limited tool in evaluating the environmental impact of pesticides and it requires a substantial amount of time, skill, motivation, and effort.

The problems associated with carcass searching following pesticide kills in agriculture have been reviewed by Mineau and Collins (1988).

Some of the factors which may bias carcass searches include scavenging by other animals, detectability of carcasses, and movement of affected animals from the study area before dying. The density of ground vegetation influences the proportion of carcasses found (Pearce 1968; Buckner 1974). Poisoning by a pesticide may result in behavioural changes (Grue and Shipley 1981) decreasing the detectability of the afflicted individuals. If affected birds do become less detectable, efforts to find them are likely to be unsuccessful (e.g. Stutzenbaker et al. 1984). Poisoning may also increase the susceptibility of exposed birds to predation and consequently may decrease the probability of them being detected before predators find them (Fowle 1965; Pearce 1967). Buckner (1974) noted that most bird casualties tend to be discovered within 24 h of spray application.

The rate at which carcasses disappear from the environment is variable (Stickel 1964). Fowle (1965) cited an experiment in which carcasses placed in the forest remained for 5-7 d. However, Balcomb (1986) found scavengers removed most carcasses placed in

agricultural fields within the first 24-h period. Therefore, unless the disappearance rate is known, carcass searches should be conducted immediately following times of expected mortality, such as immediately following a spray. But delayed mortality due to pesticide-induced anorexia, cold stress or other environmental factors would be unlikely to be synchronized among birds, rendering carcass searches a relatively ineffective monitoring technique.

Stutzenbaker et al. (1984) conducted a field study to document the effectiveness of search crews in locating waterfowl carcasses in wetland habitat and to determine the length of time carcasses persist after death. An eight-man crew, consisting of experienced wildlife professionals, searched a 40-ha site for sick or dead birds where, unknown to them, 100 duck carcasses had been placed 30 min earlier. Fifty of the birds were placed in typical escape cover, and the remaining birds were randomly placed in completely exposed locations. Searchers found none of the birds placed in locations with overhead cover. They found only six (12%) of the birds which had been left completely exposed. That study demonstrates that extensive waterfowl mortality could occur with few carcasses being found by search crews, indicating that casual searches would almost invariably result in negative findings. The portion of their study having to do with carcass longevity demonstrates that low-density waterfowl die-offs would be difficult, if not impossible, to detect because carcasses disappear quickly. Only large-scale losses are likely to be detected when mortality exceeds the capability of predators and scavengers to remove the evidence.

The biases discussed above affect the detection of carcasses and, ultimately, contribute to underestimates of mortality. Some authors (e.g. Pearce 1968; Heinz et al. 1979; Stutzenbaker et al. 1984) consider that the detection of even a small number of birds is reason enough to suspect major pesticide-induced mortality in view of the difficulty of locating carcasses, and the rapidity with which they disappear (Davis 1970; Stutzenbaker et al. 1984; Balcomb 1986). The usefulness of carcass searches in determining the impact of a pesticide is thus relatively limited since negative findings probably do not negate the possibility of widespread mortality having occurred. The use of retrieving dogs could potentially improve the efficiency of carcass searches.

3.6.2 Population Censuses

3.6.2.1 Singing-bird surveys

Traditionally, the most common method employed to measure the impact of spraying insecticides in forests was monitoring the avian population response by comparing pre- and post-spray singing activity of territorial males (e.g. Finley 1965; McLeod 1967; Fowle 1972; Moulding 1976; Pearce et al. 1976, 1979a; Buckner and McLeod 1977; Bart 1979; DeWeese et al. 1979; Richmond et al. 1979). A decrease in singing activity presumably indicated an impact but frequently ambiguous results were attained because of numerous complicating factors (Pearce et al. 1979a). Those factors include spring migration, non-uniform spray application, emigration in response to spray, takeover of resident males' territories by immigrating or resident non-singing males ("floaters"), out-of-area feeding in response to pesticide-induced decreases in food availability, and variability in the number of birds detected relating to factors other than spray (e.g. weather, time of day, or different stages of the nesting cycle) (Moulding 1976; Pearce et al. 1979a; Richmond et al. 1979; Ralph and Scott 1981; Lehoux et al. 1982).

Singing-bird surveys should be conducted along transects or by making point counts in large operational spray blocks. Small plots are subject to bias from movement of birds in and out, non-uniformity of spray distribution, and difficulty of statistical analysis because of small sample size (Peakall and Bart 1983). Because of the many inherent methodological problems associated with census techniques, survey results are best used only as an index of abundance, not to indicate absolute densities of birds (Ralph and Scott 1981; Verner 1985).

The most significant problem associated with singing-bird surveys is that depression of singing activity may not be directly related to mortality or emigration. There are many reports of reduced activity, particularly song production, during singing-bird surveys following applications of OPs or carbamates (Finley 1965; Giles 1970; Fowle 1972; Pearce et al. 1976; Bart 1979). Grue and Shipley (1981) found a significant reduction in activity, including singing, of caged European Starlings following a single dose of dicrotophos. They suggested that census techniques by themselves may be an inadequate means of assessing the acute impact of forest insecticides because decreases in singing activity cannot be attributed solely to mortality or emigration.

Singing-bird surveys, in conjunction with carcass searches, were the method of choice in the 1960s and early 1970s for detecting major impacts such as widespread adult mortality from phosphamidon and fenitrothion spraying. It is increasingly obvious that such methods are not capable of detecting impacts such as nestling mortality, prevention of breeding attempts, and low levels of adult mortality (Peakall and Bart 1983). Although it is generally accepted that birds eventually recover from non-lethal doses of OP insecticides, sublethal effects associated with OP exposure generally received little attention. Present use of singing-bird surveys should be restricted to determining that a new insecticide or spray regime does not cause widescale mortality or should be used as part of a monitoring program to detect spray applications in excess of recommended dosage rates (Peakall and Bart 1983). Techniques capable of detecting more subtle effects are required.

3.6.2.2 Territory mapping

Measuring forest bird populations by territory mapping or determining the fate of individuals is a potentially useful method of studying the impact of insecticide spraying. Identification of territorial patterns and demonstration of absence of migratory fluctuations prior to spray are necessary (Pearce 1968). If, however, individuals are not marked (e.g. Holmes et al. 1981; Kingsbury and McLeod 1981; Millikin 1987), the data are difficult to assess because of the possibility that effects may be masked by the immigration and replacement of territory holders by "floaters" (Mineau and Peakall 1987).

Territory mapping and the capture and marking of individual birds are both extremely labour-intensive procedures and, therefore, are generally limited to small plots which are seldom replicated. Thus, territory mapping is often subject to the same biases associated with the use of small plots, as noted above, for singing-bird surveys. The International Bird Census Committee recommends a minimum plot size of 10-30 ha in closed habitats and 40-100 ha in open habitats for census mapping work (Svensson 1970). The cost and amount of labour involved in relation to the often small data set obtained may prohibit that method from wide use (Heinz et al. 1979; Richmond et al. 1979).

3.6.2.3 Netting studies

The netting of birds to mark and recapture individuals (without reference to territory mapping) is unlikely to be a valuable method in itself because it is difficult to distinguish netting biases from

pesticide-related effects. Examination of only those individuals or species which are most easily trapped will underestimate the impact on species or age classes more difficult to sample (Heinz et al. 1979). Nets are generally not at sufficient heights to adequately sample upper-canopy birds (e.g. Lehoux et al. 1982). Nets may tend to capture active, healthy individuals; they would be less likely to catch birds whose sublethal exposure resulted in reduced activity and movement. Also, sample sizes for any particular species or age class are often extremely small, making statistical analysis difficult or necessitating analysis without making species distinctions and without reference to species-specific dispersal patterns of juveniles (e.g. Lehoux et al. 1982; Millikin 1987).

Netting birds has also been used to measure reproductive success (e.g. Millikin 1987). However, if the peak hatch date is a week or more after the insecticide application it may be difficult to construct a causal relationship between a spray application and the number of juveniles captured.

Results from netting studies are frequently given as number of birds captured per unit netting effort (i.e. per 100 net hours). Unfortunately, the method used in netting studies is often inadequately described, making evaluation difficult. Biases may arise from the location and mesh size of nets, weather conditions, species "catchability" (or net shyness), hours of operation, net attendance, and comparability of control and experimental habitats.

3.6.3 Nesting Studies

Nesting studies and measures of reproductive success provide the most effective and sensitive measures of insecticide spray impacts because adverse effects on both the parents and young can be determined (Richmond et al. 1979; Grue and Shipley 1981). Young, altricial songbirds can be affected directly or indirectly if a parent bird is sick or dead and, therefore, cannot care for its eggs or young. Nesting studies are extremely labour intensive and are often species-specific, for logistical reasons. Consequently, nesting studies of free-living birds tend to be feasible only on small plots and are subject to the same small plot biases described above. However, Holmes (1988) examined reproduction in a laboratory study of caged Zebra Finches in which many of the problems inherent in field studies could be eliminated. Unfortunately, laboratory studies do not closely simulate aerial applications in the field. Both approaches may be necessary to adequately address the effects of insecticide spraying on reproduction in birds.

Young altricial birds are suspected of being particularly susceptible to ChE-inhibiting insecticides because of naturally low levels of brain ChE (Grue et al. 1981; Grue and Shipley 1984) in combination with low body weight (Peakall and Bart 1983). No other method currently in use provides the opportunity to measure the impact of forest spraying on young birds. In addition, nesting studies, in conjunction with behavioural observations, can be used to estimate the effects of pesticide-induced depression of food supply on adult foraging behaviour and nestling growth and survival.

3.6.4 Observations of Abnormal Behaviour

Early studies of fenitrothion effects on forest birds using carcass searches and singing-bird surveys were replete with observations of birds which behaved abnormally (e.g. Pearce 1967, 1968; Buckner 1974; NRCC 1975). Pearce (1967) described birds as appearing to behave abnormally following fenitrothion spraying. Birds engaged in constant bill wiping and demonstrated difficulty maintaining a perching or upright position. They showed a reluctance to fly, allowed observers to approach closely, and exhibited an inability to escape capture by observers using their hands. Zinkl et al. (1979) and Richmond et al. (1979) observed profuse salivation, inability to maintain a perching position, and tremors in birds following aerial application of acephate. In most studies, however, observers reported abnormal behaviour without describing it, while in more recent investigations (e.g. Buckner and McLeod 1977; Busby et al. 1981; Kingsbury and McLeod 1981; Strong and Wells 1987; Fudge et al. 1987) there was simply a statement of no observation of abnormal behaviour. The presence or absence of abnormal behaviour should not be used as a criterion to determine whether forest birds have been poisoned by insecticide spraying unless "abnormal" behaviour is clearly defined.

There also appears to be an assumption that all abnormal behaviour is conspicuous. While that may be true when acute poisoning occurs, recent evidence indicates that birds may actually become less active following sublethal exposure (Grue and Shipley 1981; Holmes 1988). Thus, if birds become quiet and secretive following spraying, the likelihood of such "abnormal" behaviour being detected is reduced.

There is a more serious objection to using a lack of observations of abnormal behaviour as evidence of no impact of forest spraying on songbirds. The relationship between the onset and duration of

clinical signs of OP poisoning and the level of brain ChE inhibition appears to be very complex (Clark 1971; Bignami et al. 1975). If the animal survives the initial exposure, it may become symptom-free within a few hours or days even though brain ChE levels remain significantly depressed or continue to decrease (Bignami et al. 1975; Costa et al. 1982). In other cases, neurological and behavioural effects may be apparent as long as the animal is exposed, although after exposure ceases, rapid recovery follows (Keith and Mulla 1966; Grue 1982). Effects may sometimes persist for relatively long periods of time (weeks or months) even though brain ChE and ACh levels returned to normal (Karczmar 1984). Current evidence suggests, therefore, that the absence of manifestations of abnormal behaviour is a poor diagnostic measure of OP poisoning or exposure.

3.6.5 Experiments with Caged Birds

Caged birds have been used to assess the impact of forest spraying on three occasions. Peterson (1969) examined brain ChE activities in caged House Sparrows (*Passer domesticus*) and Ring-necked Pheasants exposed to fenitrothion applied at a dosage rate of 420 g a.i./ha, in a forest clearing. Because most House Sparrow brains had significantly elevated ChE activity, the results were considered invalid. However, over half of the wild birds exhibited levels of inhibition over 20%. Findlay et al. (1974) exposed caged Japanese Quail to a 280 g a.i./ha spray of fenitrothion. Birds exposed in open areas were more affected than those under the forest canopy, as determined by brain and serum ChE activity measurements, although the overall degree of ChE inhibition was relatively small.

In a third study, Holmes (pers. comm.) exposed caged White-throated Sparrows to an aerial application of 280 g a.i./ha fenitrothion. (Details of the spray application are given in Millikin 1987.) At each of three sampling periods, 8 h, 28 h, and 76 h post-spray, five birds were sacrificed. Only one bird, in the 28-h sample, had a brain ChE value depressed by more than 20% from control. However, a comparison of spray cards placed inside and outside the cages revealed that deposit (g a.i./ha) inside the cages was only 63% of deposit measured outside, and that the median droplet diameter inside the cages was only about half that outside the cages. That provided direct evidence that the cage intercepted a significant proportion of the larger droplets.

Caged-bird studies may be useful if the selection of species is more representative of the small passerines most vulnerable to the sprays and if the cage

environment more closely simulates the natural setting (Mineau and Peakall 1987). However, until the problem of cage interception of the spray is resolved that technique cannot be recommended.

3.6.6 Brain Cholinesterase Activity Assessments

Determination of brain ChE activity has advantages over many other methods used to assess the impact of insecticides on birds. It relates directly to the physiological mode of action that causes mortality or sublethal effects (Peakall and Bart 1983).

In diagnosing exposure to OPs, measures of blood (red blood cell [RBC], and plasma) ChE activity apparently are not as reliable as measures of brain ChE activity. The preservation, preparation, and assay procedures are more complex; and the biological significance of depression is less well understood than that of brain ChE (Hill and Fleming 1982). Under normal conditions, RBC and plasma ChE levels are more variable than brain ChE levels. Little is known regarding their function (Kutty 1980), and, according to some authors, there appears to be no well-established relationship between the level of inhibition of RBC and plasma ChE and that of brain ChE (Frawley et al. 1963; Clark 1971; Fleming 1981). However, in a recent study Holmes (1988) found significant relationships between doses of fenitrothion to caged Zebra Finches and both brain and plasma ChE activity. A behavioural parameter (hopping) correlated well with plasma ChE activity, suggesting that plasma ChE determinations may be more useful in avian impact assessments than previously thought. Also, Farage-Elawar and Francis (1988) reported a good correspondence between whole blood AChE (acetylcholinesterase - the other ChEs in blood having been selectively inhibited) activity in a small sample of young domestic chicks treated orally and dermally with fenitrothion.

Determination of brain ChE activity has been used to assess the impact of OP and carbamate insecticide spraying of forests on wild birds in numerous studies (e.g. Peterson 1971, 1976; Zinkl et al. 1977, 1979, 1980; Julin and Gramlich 1978; DeWeese et al. 1979; Richmond et al. 1979; Busby et al. 1981, 1982, 1983a, 1983b, 1987a, 1987b; Fudge et al. 1987, 1988; Strong and Wells 1987; Busby and Pearce 1988). Generally accepted criteria based on the degree of brain ChE depression have been used for diagnostic purposes in implicating OP-induced mortality (greater than 50% depression) and in detecting sublethal exposure (greater than 20% depression) in

birds (Ludke et al. 1975). In a modification of those guidelines, Zinkl et al. (1980) considered using the lower of 2 SD and 20% less than the mean control activity as indicative of exposure.

The degree of brain ChE depression in relation to dosage level of OPs varied highly among individuals of the same species (Fleming 1981; Fleming and Bradbury 1981; Fleming and Grue 1981) indicating a wide range in sensitivity to OPs, even under controlled conditions. Thus, it is unlikely that uniform brain ChE depression would be observed in wild birds after aerial application of OPs because both exposure and individual response to the same exposure would vary.

Some of the limitations and biases inherent in brain ChE determinations for assessing insecticide spray impacts on forest birds received little attention until recently (Peakall and Bart 1983; Mineau and Peakall 1987). Brain ChE determinations require the collection of individuals and, therefore, the impact assessment technique is subject to biases from collection methods (Peakall and Bart 1983; Mineau and Peakall 1987). As discussed above with reference to singing-bird surveys, birds dosed with ChE inhibitors tend to be less active, to sing less, and to flush less readily (Grue and Shipley (1981). Such behavioural aberrations in response to OP exposure would bias collection of birds towards healthier individuals, while heavily exposed individuals would be less likely to be collected (Busby et al. 1981; Peakall and Bart 1983; Mineau and Peakall 1987). A bias towards primarily healthy individuals is probably only minimally offset by collection of some individuals more conspicuous as a result of uncoordinated movements (Mineau and Peakall 1987). Use of 50% brain ChE inhibition as a guideline indicative of a potentially lethal exposure may be inadequate in measuring the hazard associated with insecticide spraying if individuals at that level of inhibition are unlikely to be sampled (Mineau and Peakall 1987).

When birds were collected over several days, maximum ChE depression occurred at varying times after spray application in different species (e.g. Zinkl et al. 1979; Busby and Pearce 1988). Thus, the average inhibition level would be reduced by the inclusion of birds sampled after substantial recovery had occurred or by inclusion of birds collected soon after spraying but before inhibition had occurred (Mineau and Peakall 1987). Data should be analyzed by collection day to minimize such biases (e.g. Busby et al. 1987a; Busby and Pearce 1988).

Busby et al. (1987b) noted control birds which exhibited brain ChE activity indicative of exposure to a ChE-inhibiting agent. Fudge et al. (1987) reported that six control birds from unsprayed forest (2 White-throated Sparrows, 1 Blackpoll Warbler [*D. striata*], 3 Yellow-rumped Warblers) had brain ChE levels of 20% less than the mean control values. However, only one bird, a White-throated Sparrow, had a brain ChE level greater than 2 SD from the control mean. Control birds with apparently inhibited ChE activity may have been exposed to ChE-inhibiting chemicals through drift or migration through sprayed areas, although random variation in natural ChE activities is most likely the reason. Because of the importance of establishing the "normal" range of control values in order to document exposure, every effort should be made to ascertain whether low control values are from birds which have in fact been exposed. One possible method would be to attempt ChE reactivation with a suitable agent (e.g. 2-PAM) (Martin et al. 1981).

Clearly, brain ChE activity determinations represent only a few fixed points in a dynamically changing continuum and provide no reference to where, on the depression-recovery curve, the bird was at the time of death or collection. A bird, for example, found to have 50% inhibition while in the recovery phase may not be equally likely to succumb to the toxin as a bird also exhibiting 50% inhibition but in the depression portion of the curve. Similarly, birds in a physiologically fit state may be less likely to die than unfit ones even though the levels of brain ChE inhibition may be equal. Thus, predictive capabilities on an individual basis are quite limited, particularly when sample sizes are small and the history of the birds is unknown. When approached in a systematic manner, with adequate post-exposure sampling, mean trends and distributions in the data should be indicative of a level of exposure valuable for a general impact assessment. As indicated above, in the case of forest songbirds after aerial application of ChE-inhibiting insecticides, samples of exposed birds probably give an underestimate of the actual mean ChE activity of the population.

The value and limitations of brain ChE determinations in indicating sublethal effects have yet to be fully assessed. Some evidence indicates a relationship between sublethal levels of brain ChE depression and detrimental effects on survival (Galindo et al. 1985) and reproduction and behaviour in birds (e.g. Grue and Shipley 1981; Grue et al. 1982b; Busby in prep., data presented in section on Reproductive and Behavioural Effects). Pertinent information is difficult to obtain; brain ChE determinations require the animal to be sacrificed,

limiting the ability to collect data on behavioural and reproductive parameters. Furthermore, the question remains whether sublethal levels of brain ChE depression indirectly affect the survival of birds in the wild by impairing their ability to avoid predators (Galindo et al. 1985), to complete migration, to withstand inclement weather conditions, and to manage many other environmental stresses.

3.7 ROUTES OF EXPOSURE

There is little quantitative information for birds on the relative importance of different routes of exposure to aerially applied forest insecticides. Potential routes include direct exposure through inhalation and dermal absorption, and secondary exposure through ingestion via preening of contaminated feathers or consumption of contaminated food.

Exposure of birds to fenitrothion is partially dependent on the persistence of fenitrothion in the environment. Contamination of the air is known to be relatively short-term following spraying, although the level of contamination varies in relation to the type of chemical, the formulation used, spray timing, and environmental factors (Sundaram 1984a). Exposure through inhalation probably occurs immediately following a spray event. Most of the fenitrothion residues on foliage of spruce and fir dissipate within 10-14 d after spraying (Yule and Duffy 1972; Sundaram 1974a), with maximum loss occurring within 5 d of application. Exposure through direct contact with contaminated foliage thus tends to occur primarily within the first 5 d following spraying. Birds are expected to be exposed to fenitrothion through ingestion of contaminated insects for only a few days because of rapid insect kill and insecticide degradation.

The potential risk from inhalation that birds face during aerial application of fenitrothion or other insecticides has not received direct examination through toxicological studies in laboratory or field. However, exposure through inhalation would be expected to be greater in birds than in other organisms of similar weight. Fresh air passes through birds' respiratory systems during both inhalation and exhalation; and they have a high respiration rate due to their relatively small size (Welty 1975). Exposure of birds to fenitrothion through inhalation would vary with activity, since the respiration rate increases during locomotory activity and during vocalizations. As well, direct contact with fenitrothion is probably greater during daylight hours when songbirds are most active.

Exposure of birds to insecticides may also occur through dermal absorption. Experiments by Fowle (1965) and Morgan (1968) indicated that dermal absorption of phosphamidon and fenitrothion either topically applied or through contact with contaminated foliage caused mortality in songbirds. Seventy-four percent of 202 birds collected after aerial spraying of trichlorfon and carbaryl had tracer dye on their feathers or their feet, or both (DeWeese et al. 1979). The feet of individuals of the ten species examined frequently had the most dye, suggesting that in birds dermal absorption may be an important route of exposure to aerially-applied insecticides. Although fenitrothion was found to have a low absolute dermal toxicity compared to 18 other OP insecticides, it did have the highest dermal toxicity index and was more toxic dermally than orally (Hudson et al. 1979). Also, Farage-Elawar and Francis (1988) found the maximum brain ChE inhibition induced by oral and dermal routes of exposure to fenitrothion to be equal. Thus, the dermal route of exposure may be particularly important in the case of fenitrothion.

Hamilton et al. (1981) considered direct contact with either the spray cloud or contaminated foliage to be the principal pathway of fenitrothion uptake in birds after spraying. They found maximum residues of fenitrothion on birds occurred 1 d after spray and found at least 95% of total fenitrothion residues on the plumage and skin of Chaffinches and Coal Tits. Also, they suggested that fenitrothion could be ingested indirectly from contaminated feathers through preening or absorbed directly through the skin.

Secondary exposure may occur through the ingestion of dead or struggling insects poisoned by insecticide spraying. Stickel (1974) considered that to be the most important route of exposure in wild birds. Many forest songbirds prey on spruce budworm, larvae and pupae (Mook 1963; Morris 1963; Gage 1968; Crawford and Jennings 1982) and, therefore, would be particularly at risk from ingestion of insecticide residues following the spray. Those species include Golden-crowned Kinglet (*Regulus satrapa*), Ruby-crowned Kinglet (*R. calendula*), Red-eyed Vireo (*Vireo olivaceus*), Cape May Warbler (*D. tigrina*), Magnolia Warbler (*D. magnolia*), Bay-breasted Warbler (*D. castanea*), Blackburnian Warbler (*D. fusca*), Tennessee Warbler (*Vermivora peregrina*), and White-throated Sparrow. There is some evidence to suggest that breeding populations of some species (e.g. Bay-breasted, Cape May, Blackburnian, and Tennessee Warbler) frequently exhibit a dramatic numerical response to high

budworm densities (Morris et al. 1958; Sanders 1970; Welsh and Fillman 1980). That raises the possibility of those species being particularly vulnerable to insecticide spraying.

In studies which have quantified the concentration of OPs and their metabolites in insects, investigators have considered the residue concentrations to be below those lethal to wildlife (McEwan et al. 1972; Hamilton et al. 1981; Stromborg et al. 1982). Exposure from the ingestion of contaminated insects may vary in relation to the magnitude of the insect kill, such that birds may consume more contaminated insects when the spray kills more insects. It is worth noting that hair-cap moss (*Polytrichum commune*) capsules, a favoured food of adult White-throated Sparrows, have consistently much higher residues than insects (D. G. Busby, Appendix 3.2). That evidence suggests that for birds including some plant matter in their summer diets, ingestion of parts of contaminated plants may be more important than consumption of insects in the intake of insecticide after spray applications.

There is no information on whether the important routes of exposure in young songbirds differ from those in adults. It is possible that nestlings would be more vulnerable to exposure from contaminated food because of their high ingestion rates relative to body size. One-day-old nestling White-throated Sparrows ate 176% of their body weight in insects while 4- and 7-d old nestlings consumed 72% and 77%, respectively, of their body weight in insects (Busby 1982a). Young songbirds are confined to the nest until they fledge and, therefore, would not be expected to be exposed through contact with contaminated foliage. They may be exposed, however, to the contaminated plumage of their parents while being brooded.

Eidt and Pearce (1986) calculated the amount of fenitrothion-contaminated foliage that a Spruce Grouse (*Dendragapus canadensis*) would need to consume to get an LD50 dose. They assumed that the sensitivity of Spruce Grouse was similar to that of the closely related Sharp-tailed Grouse (*Tympanachus phasianellus*), with an LD50 of 53.4 mg/kg. To achieve a similar level of exposure a 550 g Spruce Grouse would have to eat 29 kg of conifer needles containing 1 ppm fenitrothion. However, as a Spruce Grouse does not generally consume more than 15% of its body weight per day in food, the bird would consume only 83 g/day of conifer needles, providing a margin of safety greater than 350 times the LD50 dose.

Crick and Spray (1987) used the residue data of Hamilton et al. (1981) to calculate the potential dose of fenitrothion that adult Coal Tits may receive from ingestion of contaminated insects and directly from their plumage. According to their calculations, adult Coal Tits may receive 7.5 mg/kg body weight of fenitrothion directly on their plumage; and they could ingest up to a maximum of 9.6 mg/kg daily in their diet. On the basis of known LD50s and LC50s of passerine species (Tables 3.1 and 3.2) weighing much more than the Coal Tit, they conclude that adult Coal Tits could potentially acquire a lethal or significant sublethal dose of fenitrothion within a few days of spraying. Crick and Spray also calculated that the potential intake of fenitrothion for 6-g and 9-g nestlings would be 21.42 mg/kg and 14.28 mg/kg, respectively, suggesting that nestling Coal Tits may be particularly vulnerable to fenitrothion spraying.

In general, the impact of aerial application of OPs is apparently greatest on upper canopy foragers (Pearce 1967, 1968; Buckner et al. 1973; Moulding 1976; Zinkl et al. 1977; Pearce et al. 1979a). The species of birds inhabiting the upper- and mid- parts of the canopy of eastern coniferous forests tend to be those that prey on spruce budworm. Upper-canopy birds would be expected to be more exposed than species found lower in the canopy. They likely inhale greater quantities of insecticide from the air because they are closer to the level of spray emission. Upper-canopy birds would come into greater contact with spray deposit because foliage in the upper canopy intercepts much of the spray.

Collection of spray deposit by the canopy may provide some protection for birds found lower down. Japanese Quail caged in areas where there was no canopy protection had a greater degree and incidence of brain ChE depression than birds under a spruce canopy (Findlay et al. 1974). However, Findlay et al. (1974) noted that although canopy protection will reduce the impact of fenitrothion upon birds in a sprayed forest, it will not provide complete protection. Some spray will always reach birds found lower in the canopy. Also, Busby and Pearce (1988) found the lower-canopy dwelling White-throated Sparrow to show the greatest response to UULV fenitrothion spraying.

After examining the occurrence of tracer dye from aerial application of trichlorfon and carbaryl on birds in Douglas fir forests, DeWeese et al. (1979) concluded that the canopy provided only limited protection to birds by intercepting spray and decreasing the amount that reached understory vegetation. However, their methods give only qualitative results (presence or absence), and the

quantity of deposit encountered by upper- canopy birds is probably greater than those in the lower canopy.

3.8 DATA SPECIFIC TO FENITROTHION USE IN FORESTRY

In this section, the impacts on birds of fenitrothion use for control of forest insect pests is briefly reviewed. It covers most of the data collected during operational or experimental applications of fenitrothion as well as a few relevant laboratory studies. A large part is devoted to the data available on brain ChE depression in field-collected samples. That widely utilized technique, despite inherent biases outlined above, is one of the best available for routine monitoring purposes (Mineau and Peakall 1987).

All of the evidence currently available on the effects of fenitrothion on reproduction and behaviour and direct mortality is reviewed and commented on. Finally, information on residues in collected birds is presented, more for the sake of completeness than for its inherent value.

It is not proposed to review the bird census information available from the various fenitrothion spray monitoring studies. As discussed above, the methods used are extremely insensitive and the results usually difficult to interpret. Peakall and Bart (1983) reviewed much of the relevant literature.

3.8.1 Brain Cholinesterase Data

3.8.1.1 Spray dosage rate

Adequate brain ChE data for songbirds collected after the application of fenitrothion are available from a total of nine separate studies, including 19 separate spray blocks or 27 separate spray events if both first and second sprays are tallied separately (Table 3.4). Of the nine, five are from New Brunswick (P.A. Pearce, unpub. data; Busby et al. 1981, 1983a, 1987a; Busby and Pearce 1988), three are from Newfoundland (Strong and Wells 1987; Fudge et al. 1987, 1988), and one is from Scotland (Hamilton et al. 1981). Two studies from Maine (Peterson 1969, 1976) are excluded from this analysis because too few birds of any one species were collected to provide meaningful results.

Looking only at the 19 first spray events (regardless of dosage rate), the proportion of collected birds exhibiting greater than 20% ChE inhibition ranged

Table 3.4 Proportions of birds having brain ChE depression $\geq 20\%$ and $\geq 50\%$ in relation to dosage rate of fenitrothion.

Dosage rate ¹ (g a.i./ha)	No. Species	Sample size		% of birds with ChE depression		Source ²
		Sprayed	Controls	$\geq 20\%$	$\geq 50\%$	
140 (1st)	3	36	60	44	6	1
140 (2nd)	3	36	60	34	3	1
210 (1st)	4	101	46	9	0	2
210 (2nd)	4	93	46	23	1	2
210 (B102)	2	39	20	23	3	3
210 (B103a)	2	36	20	31	6	3
210 (B103b)	2	37	20	19	3	3
210 (B103c)	2	32	20	66	34	3
210 (B107)	3	46	30	33	4	3
210 (1st, B105)	3	77	30	74	35	3
210 (2nd, B105)	3	79	30	72	19	3
140	4	83	47	66	18	4
210 (1st)	4	80	47	58	38	4
210 (2nd)	4	83	47	58	13	4
210 (1st)	3	27	43	76	55	5
210 (2nd)	3	26	43	76	20	5
210 (1st)	1	36	14	25	8	6
210 (2nd)	1	27	14	33	0	6
210 (1st)	1	12	16	58	0	7
210 (2nd)	1	30	16	50	7	7
280 (RA plot)	5	59	47	16	5	8
280 (B&N plot)	5	63	47	30	5	8
300	2	33	11	61	9	9
300	2	95	14	53	10	9
350	1	6	43	67	33	5
420 (1st)	1	20	10	95	25	7
210 (2nd)	1	18	10	83	6	7

¹ (1st) or (2nd) = 1st or 2nd spray. B = Block #.

² (1) Pearce, unpublished data; (2) Busby et al. 1987a; (3) Fudge et al. 1988; (4) Busby and Pearce 1988; (5) Fudge et al. 1987; (6) Strong and Wells 1987; (7) Busby et al. 1983a; (8) Busby et al. 1981; (9) Hamilton et al. 1981. Results from sources (5) and (6) recalculated separately for 1st and 2nd spray from data in appendices. Data from source (9) obtained from graphs and source (4) from Table 6.

between 9% and 95%, with a median value of 53%. The proportion of birds inhibited more than 50% ranged between 0% and 55%, with a median of 8%. Considering all spray events, all spray blocks had some birds inhibited more than 50% regardless of spray dosage rate.

Despite the fact that the highest dosage rate (420 g a.i./ha) resulted in the highest proportion of birds being "exposed," the data do not indicate a strong relationship between the number of inhibited birds at either 20 or 50% and the dosage rate. It is believed that is primarily a result of the strong imbalance in the data (i.e. an over representation of the 210 g a.i./ha rate) as well as the very large degree of variability within any one spray dosage rate. Mineau and Peakall (1987), working with a more restricted data set for fenitrothion spray events and with a more balanced data set for another pesticide, acephate, showed that the proportion of birds deemed to have been exposed (20% inhibited) did increase with application dosage rate. However, the mean inhibition level showed no clear relationship with application dosage rate. They attributed those findings to a collection bias in favour of the least inhibited birds and to the very great difficulty in sampling severely inhibited birds because those were either dead or sick and well hidden.

The wide variation in the biological response to the same dosage rate (e.g. 210 g a.i./ha) is striking. Many factors affect the total amount or pattern of spray deposit of a particular spray application. In addition, results are frequently combined for a number of species varying appreciably in their sensitivity to the insecticide, their vulnerability, and the routes of exposure. In the following sections, a few of the factors probably contributing to the variation in the magnitude of the recorded brain ChE depression of birds among spray events are examined. Those are forest structure and topography, spray method, and the several biases inherent in collecting representative samples of birds. There is also the need to consider the confounding effect of repeat spray applications - the norm for some of the ongoing insect control programs.

3.8.1.2 Forest structure and topography

The influence of forest structure and topography on spray impacts on birds has received little attention. However, the data presented in Table 3.4 indicate that those factors may be relevant to exposure of birds to aerially applied insecticides. The proportion of birds with 20% or more brain ChE depression is strikingly similar for first and second applications of 210 g a.i./ha within each study (i.e. within each

spray block) but ranged from 9 to 76% for first sprays and from 23 to 76% for second sprays between studies (Table 3.4). That similarity of response within each spray block and different response between blocks provides strong evidence that the structure of the forest being sprayed and the topography may play an important role in determining the effects the spray will have on the avifauna.

3.8.1.3 Influence of spray method

It is increasingly obvious through years of aerial use of pesticides in both agriculture and forestry that the method of delivery is of equal or greater importance than the pesticide dosage rate (McCracken 1980). Historically, New Brunswick has used a bewildering array of spray regimes, which changed frequently from year to year (Pearce and Peakall 1977). The spray history of a particular area is unlikely to be precisely the same as that of another area because of changes in the insecticides used, dosage rates, combinations of treatments, and sequence of applications (Pearce and Peakall 1977). The variety of spray regimes hampered the scientific investigation of environmental impacts making it difficult to isolate the relative importance of any single variable.

Relatively few studies specifically addressed how spray method may modify the impact of forest insecticides on songbirds. Those studies primarily focused on spray timing, and the type of aircraft and spray equipment used for spraying, and are discussed below.

Pearce (1968) presented evidence from singing-bird surveys indicating that morning sprays have a greater effect on forest birds than evening sprays. Although results were qualified, Busby et al. (1981) found that brain ChE depression was greater in birds after morning sprays than after an evening spray. Evening sprays might be expected to have less impact than those early in the morning because of the diurnal activity patterns of songbirds. Birds tend to be more active in the early morning and, therefore, more exposed to the sprays.

Contamination of the air mass within the forest varies between morning and evening sprays (Sundaram 1984a). Concentrations of fenitrothion and the carbamate aminocarb were higher and persisted longer in the air mass within the forest following evening sprays than after morning sprays. Sundaram suggested that volatilization from forest substrates was related to air temperature, on warm evenings being greater than on cool mornings.

Sundaram's (1984b) evidence suggests either that inhalation is not an important exposure route for birds or that exposure through inhalation is much reduced when birds are less active. Another possibility is that when birds are less active there is less chance of coming in contact with contaminated foliage. Greater loss of fenitrothion from contaminated foliage through volatilization while the birds are inactive may reduce subsequent exposure.

On the basis of singing-bird surveys, Pearce et al. (1979a) concluded that the OP phosphamidon was more toxic when sprayed by TBM aircraft than by DC-6 aircraft. Also, fenitrothion sprays had a greater impact on songbirds when applied by TBM aircraft than at the same dosage rate but different formulations by a DC-6 aircraft (Pearce et al. 1979a).

Recently, New Brunswick tested UULV spray technology. Although different formulations were used, the impact of UULV spray techniques on forest songbirds was examined in two studies. Busby et al. (1987a) concluded that the impact, based on brain ChE assessment, on forest birds from the formulation used in 1986 was less than that observed using conventional ULV spraying methods. In contrast, the effect on forest songbirds of 1987 spraying operations was relatively severe in relation to the previous year's UULV study and past studies in which the impact of ULV methodology was examined (Busby and Pearce 1988). Rotary atomizers were used in both investigations. Cessna aircraft were used in 1986 when a relatively minor impact was detected while in 1987 TBM aircraft were used, having a greater impact on songbirds. Unfortunately, not only did aircraft type differ but also spray formulations varied. It is not clear which features of aircraft type might be responsible for differential effects on birds. Different flight altitudes, speeds, turbulence, and ability to adjust to uneven terrain may all be important. Each may influence the emission and deposition characteristics of the spray and thus the effects on birds.

Both ULV and UULV application methods require turbulence to give fine droplets the momentum to impact on the canopy (Joyce et al. 1981). Larger aircraft (e.g. TBM, DC-6) create greater air turbulence than small aircraft (e.g. Cessna) and, therefore, the former would be more efficient at driving fine spray droplets into the canopy. That may be modified by the height of the aircraft as it flies over forest during spraying operations. Both DC-4 and DC-6 aircraft are fitted with sophisticated electronic guidance systems and may be less subject

to navigational errors, resulting in partial or total overswathing, than the visually guided TBM aircraft.

Pearce (1968) suggested that sprays applied in a finer droplet size spectrum have a greater impact on songbirds than coarser (i.e. larger) spray droplets. Unlike fine spray droplets, coarse droplets have a greater tendency to impinge on the ground than on the forest canopy (Joyce et al. 1977; Barry 1984). Consequently, exposure of birds to contaminated foliage or through inhalation may be reduced with large spray droplets. On the other hand, larger droplets are likely to result in more dermal absorption when those droplets directly impinge on the birds (Pope and Ward 1972). Also, the impingement of larger droplets on moving birds is greater than that of small droplets, the latter being more affected by the turbulence and air currents generated by a live target (Pope and Ward 1972). Boom and nozzle equipment produces a coarser droplet size spectrum than rotary atomizers. Birds exhibited greater ChE depression when fenitrothion was sprayed with boom and nozzle equipment than when it was applied with rotary atomizers at 280 g a.i./ha (Busby et al. 1981). Busby et al. (1983b) also observed a greater impact on forest birds when TBM aircraft using boom and nozzle equipment sprayed aminocarb than when it was applied by rotary atomizer-equipped Cessna 188 Ag-Truck aircraft. Results from both studies were qualified because potential confounding factors such as timing of applications, differing formulations, and terrain variability may have influenced exposure of birds to the insecticide.

Results from the nine studies listed in Table 3.4 are grouped according to aircraft type and spray equipment and the timing (morning or evening) of the spray application noted in Table 3.5. Few obvious trends are apparent except that Cessna aircraft, regardless of atomizer type and spray dosage rate, appear to cause the least impact. The variability in avian responses in relation to dosage rate is so great that it is not surprising that differences in impacts between morning and evening sprays cannot be detected.

Fudge et al. (1988) first examined the impact of fenitrothion spraying over a number of replicates when aircraft type, spray equipment, formulation, and dosage rate were kept the same. The variability in avian response to fenitrothion, when the above factors were constant, was quite large (Table 3.5) and is probably the consequence of a wide variability in the amount and pattern of spray deposit. Present methods of estimating the amount and pattern of spray deposit on foliage or artificial targets indicate

Table 3.5 Proportions of birds having brain ChE depression $\geq 20\%$ and $\geq 50\%$ in relation to dosage rate of fenitrothion, aircraft type, spray equipment, and timing of spray applications.

Dosage rate ¹ (g a.i./ha)	Time of day ²	Aircraft	Spray equipment ³	% of birds with ChE depression		Source ⁴
				$\geq 20\%$	$\geq 50\%$	
210 (1st)	M	Cessna	RA	9	0	2
210 (2nd)	M	Cessna	RA	23	1	2
280	M	Cessna	RA	16	5	8
280	M	Cessna	B&N	30	5	8
210 (1st)	E	DC-4	B&N	76	55	5
210 (2nd)	M/E	DC-4	B&N	76	20	5
350	M	DC-4	B&N	67	33	5
210 (1st)	M	DC-6	B&N	25	8	6
210 (2nd)	M	DC-6	B&N	33	0	6
210 (B102)	E	DC-6	B&N	23	3	3
210 (B103a)	M	DC-6	B&N	31	6	3
210 (B103b)	M	DC-6	B&N	19	3	3
210 (B103c)	M	DC-6	B&N	66	34	3
210 (B107)	M	DC-6	B&N	33	4	3
210 (1st, B105)	M	DC-6	B&N	74	35	3
210 (2nd, B105)	E	DC-6	B&N	72	19	3
210 (1st)	M	TBM	B&N	58	0	7
210 (2nd)	M/E	TBM	B&N	50	7	7
420 (1st)	M	TBM	B&N	95	25	7
210 (2nd)	M/E	TBM	B&N	83	6	7
140 (1st)	M	TBM	B&N	44	6	1
140 (2nd)	M	TBM	B&N	34	3	1
140	M	TBM	RA	66	18	4
210 (1st)	M	TBM	RA	58	36	4
210 (2nd)	M	TBM	RA	58	13	4
300	?	Porter	RA	53	10	9
300	?	Porter	RA	61	9	9

¹ (1st) or (2nd) = 1st or 2nd spray. B = Block #.

² M = morning; E = evening.

³ RA = rotary atomizer; B&N = boom and nozzle.

⁴ (1) Pearce, unpublished data; (2) Busby et al. 1987a; (3) Fudge et al. 1988; (4) Busby and Pearce 1988; (5) Fudge et al. 1987; (6) Strong and Wells 1987; (7) Busby et al. 1983a; (8) Busby et al. 1981; (9) Hamilton et al. 1981.

that both the quality and quantity of spray are highly variable within and between spray events. Fudge et al. (1988) noted that the measured variation of foliage deposit could not explain the variability in the proportion of brain ChE-inhibited birds, nor could the level of brain ChE inhibition be explained

by the level of foliage deposit. Sampling may not, however, have coincided with the period of peak inhibition (see below).

Data from the studies listed in Table 3.5 were recalculated and tabulated for each of four bird

Table 3.6 Proportions of White-throated Sparrows with $\geq 20\%$ and $\geq 50\%$ brain ChE depression in relation to dosage rate of fenitrothion, aircraft type, and spray equipment.

Dosage rate ¹ (g a.i./ha)	Aircraft	Spray equipment ²	N	% of birds with ChE depression		Source ³
				$\geq 20\%$	$\geq 50\%$	
210 (1st)	Cessna	RA	23	0	0	1
210 (2nd)	Cessna	RA	20	5	0	1
280	Cessna	RA	16	6	6	2
280	Cessna	B&N	13	31	8	2
210 (1st)	DC-4	B&N	19	79	58	3
210 (2nd)	DC-4	B&N	21	90	24	3
350	DC-4	B&N	6	67	33	3
210 (1st)	DC-6	B&N	31	25	8	4
210 (2nd)	DC-6	B&N	27	33	0	4
210	DC-6	B&N	18	6	0	5
210	DC-6	B&N	19	16	0	5
210	DC-6	B&N	21	24	5	5
210	DC-6	B&N	21	33	5	5
210	DC-6	B&N	18	56	28	5
210 (1st)	DC-6	B&N	27	52	11	5
210 (2nd)	DC-6	B&N	32	44	6	5
210 (1st)	TBM	B&N	12	58	0	6
210 (2nd)	TBM	B&N	30	50	7	6
420 (1st)	TBM	B&N	20	95	25	6
210 (2nd)	TBM	B&N	18	83	6	6
140 (1st)	TBM	B&N	12	50	8	7
140 (2nd)	TBM	B&N	12	58	0	7
140	TBM	RA	23	83	26	8
210 (1st)	TBM	RA	32	77	55	8
210 (2nd)	TBM	RA	31	86	33	8

¹ (1st) or (2nd) = first or second spray application.

² RA = rotary atomizer; B&N = boom and nozzle.

³ (1) Busby et al. 1987a; (2) Busby et al. 1981; (3) Fudge et al. 1987; (4) Strong and Wells 1987; (5) Fudge et al. 1988; (6) Busby et al. 1983a; (7) Pearce, unpublished data; (8) Busby and Pearce 1988.

species to examine brain ChE depression in relation to aircraft type and spray atomization system. As previously discussed, Cessna aircraft, regardless of type of spray equipment, had less of an impact on White-throated Sparrows than other types of aircraft (Table 3.6). It is important to note that potentially lethal brain ChE depression in White-throated Sparrows was sometimes greater at dosage rates of 210 g a.i./ha than of 420 g a.i./ha. It is not known whether that is an indication of high variability between spray events or whether lethality in the 420 g a.i./ha spray removed the most affected birds from the population, thereby biasing the results towards the least affected birds. The latter may be more probable in that particular case because severe effects on territoriality and reproduction were observed (see sub-section on Reproductive and Behavioural Effects). Although data were available only from studies of sprays by Cessna and TBM aircraft, trends similar to that observed in White-throated Sparrows were noted in Tennessee, Magnolia, and Bay-breasted Warblers (Tables 3.7-3.9).

The data indicate that the avian impact of fenitrothion spraying is extremely variable even when factors such as species, aircraft type, spray

equipment, and timing of sprays are accounted for. The fact that a dosage rate of 140 g a.i./ha can have as pronounced effects on birds as 210 g a.i./ha (Table 3.4, source 4) is the most important finding of the analysis. It is evident, therefore, that all operational dosage rates of fenitrothion used in Canadian forestry (140-280 g a.i./ha) have the potential to cause significant negative effects on forest songbirds.

3.8.1.4 Collection bias

The net effect of one collection bias, namely the differential ability to sample birds with brain ChE inhibited to various degrees, is underestimation of the spray impact on the bird community, as noted above. Most studies listed in Table 3.4 provided scant information on collection methods and consequently collection bias can only be examined in reference to the number and timing of post-spray collections of birds. Mineau and Peakall (1987) suggested that the magnitude of ChE depression may be underestimated by including birds not yet affected, or birds recovering from ChE exposure, in calculations of mean brain ChE depression. Only six of the nine studies provided data on ChE depression in relation to time after spray. Their findings are

Table 3.7 Proportions of Tennessee Warblers with $\geq 20\%$ and $\geq 50\%$ brain ChE depression in relation to dosage rate of fenitrothion, aircraft type, and spray equipment.

Dosage rate ¹ (g a.i./ha)	Aircraft	Spray equipment ²	N	% of birds with ChE depression		Source ³
				$\geq 20\%$	$\geq 50\%$	
210 (1st)	Cessna	RA	26	7	0	1
210 (2nd)	Cessna	RA	28	36	0	1
280	Cessna	RA	22	9	4	2
280	Cessna	B&N	23	22	0	2
210 (1st)	TBM	RA	31	62	38	3
210 (2nd)	TBM	RA	30	50	0	3
140	TBM	RA	21	67	19	3
140 (1st)	TBM	B&N	12	25	0	4
140 (2nd)	TBM	B&N	11	9	0	4

¹ (1st) or (2nd) = first or second spray application.

² RA = rotary atomizer; B&N = boom and nozzle.

³ (1) Busby et al. 1987a; (2) Busby et al. 1981; (3) Busby and Pearce 1988; (4) Pearce, unpublished data.

Table 3.8 Proportions of Magnolia Warblers with $\geq 20\%$ and $\geq 50\%$ brain ChE depression in relation to dosage rate of fenitrothion, aircraft type, and spray equipment.

Dosage rate ¹ (g a.i./ha)	Aircraft	Spray equipment ²	N	% of birds with ChE depression		Source ³
				$\geq 20\%$	$\geq 50\%$	
210 (1st)	Cessna	RA	25	12	0	1
210 (2nd)	Cessna	RA	25	16	4	1
280	Cessna	RA	9	33	11	2
280	Cessna	B&N	16	31	6	2
210 (1st)	TBM	RA	21	47	41	3
210 (2nd)	TBM	RA	32	38	14	3
140	TBM	RA	19	37	16	3
210 (1st)	DC-4	B&N	3	100	66	4
210 (2nd)	DC-4	B&N	1	0	0	4

¹ (1st) or (2nd) = first or second spray application.

² RA = rotary atomizer; B&N = boom and nozzle.

³ (1) Busby et al. 1987a; (2) Busby et al. 1981; (3) Busby and Pearce 1988; (4) Fudge et al. 1987.

Table 3.9 Proportions of Bay-Breasted Warblers with $\geq 20\%$ and $\geq 50\%$ brain ChE depression in relation to dosage rate of fenitrothion, aircraft type, and spray equipment.

Dosage rate ¹ (g a.i./ha)	Aircraft	Spray equipment ²	N	% of birds with ChE depression		Source ³
				$\geq 20\%$	$\geq 50\%$	
210 (1st)	Cessna	RA	27	15	0	1
210 (2nd)	Cessna	RA	20	25	5	1
280	Cessna	B&N	7	57	14	2
210 (1st)	TBM	RA	31	40	15	3
210 (2nd)	TBM	RA	33	57	5	3
140	TBM	RA	20	75	10	3
140 (1st)	TBM	B&N	12	58	8	4
140 (2nd)	TBM	B&N	12	25	8	4

¹ (1st) or (2nd) = first or second spray application.

² RA = rotary atomizer; B&N = boom and nozzle.

³ (1) Busby et al. 1987a; (2) Busby et al. 1981; (3) Busby and Pearce 1988; (4) Pearce, unpublished data.

given in Table 3.10.

The timing of maximum brain ChE depression appears to vary, even at similar spray dosage rates, among sprays, among studies, and among species (Table 3.10). The onset of brain ChE depression after fenitrothion spraying appears to vary as well. Busby and Pearce (1988) and Fudge et al. (1988) found that birds of some species did not have significant brain ChE depression when collected on the day of spray application while individuals of other species did. Zinkl et al. (1979, 1980) reported similar findings in brain ChE activity of forest songbirds after an aerial spray of acephate at 560 g a.i./ha. Although sample sizes tended to be very small, birds of some songbird species were not inhibited on the day of spray application while others were. Maximum ChE depression occurred 3-15 d after exposure to acephate. However, acephate may be a special case among even OPs because of the possibility that it breaks down to more potent cholinergic agents (Zinkl et al. 1981).

In the New Brunswick and Newfoundland studies (e.g. Busby et al. 1983a, 1987a; Busby and Pearce 1988; Fudge et al. 1988), maximum brain ChE depression in birds of some species may not have been detected because sampling was not continued long enough (only 2-3 d) to indicate that recovery had occurred. Data from the study by Hamilton et al. (1981) indicate that maximum depression in some species may not occur until much longer than 3 d after a spray event. It is thus likely that in the New Brunswick and Newfoundland investigations the impact of fenitrothion spraying was underestimated because maximum ChE depression may not have been represented in the birds sampled. In addition, the findings of those studies frequently had no reference to collection day. The overall impact assessment included birds not exhibiting brain ChE depression, as on the day of spray, and possibly biased the findings so that the effect of spraying was underestimated. Table 3.11 shows how such a bias can result in an underestimation of the effects of fenitrothion spraying.

Table 3.10 Maximum brain ChE depression in birds in relation to day collected after sprays of fenitrothion.

No. species	Dosage rate (g a.i./ha)	Days sampled after spray	Day of maximum brain ChE depression	Source ¹
3	140 (1st)	1, 6	1	1
3	140 (2nd)	1, 6	6	1
1	210 (1st)	0, 1, 2	0	2
1	210 (2nd)	0, 1, 2	2	2
1	420 (1st)	0, 1, 2, 3	1	2
1	210 (2nd)	0, 1, 2, 3	3	2
4	210 (1st)	0, 1, 3	3	3
4	210 (2nd)	0, 1, 3	3	3
4	140	1, 3	3	3
3	210 (1st)	0, 1, 3	1	4
3	210 (2nd)	0, 1, 3	3	4
2	210 (2nd)	1, 2	2	5
2	300	1, 4, 11	1-11	6
2	300	1, 2, 7, 21	1-7	6

¹ (1) Pearce, unpublished data; (2) Busby et al. 1983a; (3) Busby and Pearce 1988; (4) Fudge et al. 1988; (5) Busby et al. 1987a; (6) Hamilton et al. 1981.

It is important to realize that ChE inhibition in an individual bird, when it is sampled after spraying, represents a minimum estimate of the maximum inhibition that that bird would have experienced. Inhibition could have been greater either before or after collecting the bird. It is unlikely that the moment of collection would coincide with the exact moment of maximum inhibition. Laboratory studies, however, are more likely to detect maximum inhibition because sampling protocols are not subject to the logistical problems that characterize field

studies. Again that leads to the conclusion that field studies are likely to underestimate the actual degree of brain ChE inhibition in the exposed bird population.

To minimize bias and facilitate assessment of the maximum impact of fenitrothion spraying on forest songbirds, results should be reported for each bird in relation to each spray application and to collection day. Attempts should be made to identify when maximum brain ChE depression occurred in birds of

Table 3.11 The bias in impact assessments from inclusion of birds collected before the onset of brain ChE depression.

Species	SPRAY 1 % of birds with ChE activity inhibited		SPRAY 2 % of birds with ChE activity inhibited	
	≥20%	≥50%	≥20%	≥50%
<i>Including Day 0 Birds</i>				
Tennessee Warbler	48	26	43	0
Bay-breasted Warbler	29	10	49	3
Magnolia Warbler	38	33	31	9
White-throated Sparrow	72	38	68	23
Total	48	26	48	8
<i>Excluding Day 0 Birds</i>				
Tennessee Warbler	62	38	50	0
Bay-breasted Warbler	40	15	57	5
Magnolia Warbler	47	41	38	14
White-throated Sparrow	77	55	86	33
Total	58	38	58	13

Data from study by Busby and Pearce (1988).

each species by determining when recovery began.

3.8.1.5 Number of spray applications

In spruce budworm and hemlock looper control, fenitrothion is usually applied operationally in two sprays of 210 g a.i./ha, a minimum 5 or 6 d apart. Data on brain ChE depression in songbirds in relation to first and second spray application were available from seven studies (P.A. Pearce, unpub. data; Busby et al. 1987a; Busby et al. 1983a; Strong and Wells 1987; Fudge et al. 1987; Busby and Pearce 1988).

As indicated above, the proportion of birds showing 20% or more brain ChE inhibition after first and second sprays is similar within studies. However, the proportion of birds showing 50% or more

inhibition after first and second sprays is quite different. There is a strong trend towards fewer birds exhibiting that level of inhibition following the second spray. In order to test the significance of that trend, the data were analyzed by the method of Mantel and Haenszel for combining a series of 2 x 2 contingency tables (Breslow and Day 1980, p. 138). Whereas the proportion of birds inhibited under and over 20% did not differ significantly between the two spray events, the proportion of birds inhibited 50% or more was significantly lower than expected following the second spray for all birds combined (Table 3.12, Chi Sq. = 20.82, D.F. = 1, P < 0.005) as well as for White-throated Sparrows alone (Table 3.13, Chi Sq. = 6.12, D. F. = 1, P < 0.025).

Table 3.12 Comparison of songbirds having $\geq 20\%$ and $\geq 50\%$ brain ChE inhibition after 1st and 2nd sprays of fenitrothion for spruce budworm and hemlock looper control.

Dosage rate ¹ (g a.i./ha)	No. Species	Sample size		% of birds with ChE depression		Source ²
		Sprayed	Controls	$\geq 20\%$	$\geq 50\%$	
140 (1st)	3	36	60 ³	44	6	1
140 (2nd)	3	36		34	3	1
210 (1st)	4	101	46	9	0	2
210 (2nd)	4	93		23	1	2
210 (1st, B105)	3	77	30	74	35	3
210 (2nd, B105)	3	79		72	19	3
210 (1st)	4	80	47	58	38	4
210 (2nd)	4	83		58	13	4
210 (1st)	3	29	43	76	55	5
210 (2nd)	3	26		76	20	5
210 (1st)	1	36	14	25	8	6
210 (2nd)	1	27		33	0	6
210 (1st)	1	12	16	58	0	7
210 (2nd)	1	30		50	7	7

¹ (1st) or (2nd) = 1st or 2nd spray. B = Block #.

² (1) Pearce, unpublished data; (2) Busby et al. 1987a; (3) Fudge et al. 1988; (4) Busby and Pearce 1988; (5) Fudge et al. 1987; (6) Strong and Wells 1987; (7) Busby et al. 1983a. Results from sources (5) and (6) recalculated separately for 1st and 2nd spray from data in appendices.

³ Within each study, one control data set is used for comparisons with 1st and 2nd sprays.

In nearly all studies, the birds were collected widely in large spray blocks (information in Fudge et al. 1988 indicates the potential overlap in collection days in two of seven blocks sampled). Thus it is unlikely that the first collection influenced the second. Rather, it is suggested that birds which were more vulnerable at the time of the first spray, (e.g. those likely to receive a greater exposure by virtue of their habits, nest location, or other factors), were selectively removed from the sample pool, thus biasing results towards less affected birds available for collection after the second spray. If that explanation is indeed correct, it means that substantial adult bird mortality goes undiscovered.

3.8.2 Reproductive and Behavioural Effects

Avian reproduction is often synchronized with periods of insect abundance (Lack 1954) and, therefore, the application of insecticides frequently coincides with the avian breeding season (Grue et al. 1983). For example, spruce budworm larviciding in New Brunswick begins mid-to-late May and runs through mid-to-late June. That overlaps considerably with the period in which most migrant songbirds return from the southern wintering grounds, set up territories, and initiate nesting. In Newfoundland,

Table 3.13 Comparison of White-throated Sparrows having $\geq 20\%$ and $\geq 50\%$ brain ChE inhibition after 1st and 2nd sprays of fenitrothion for spruce budworm and hemlock looper control.

Dosage rate ¹ (g a.i./ha)	Sample size		% of birds with ChE depression		Source ²
	Sprayed	Controls	$\geq 20\%$	$\geq 50\%$	
140 (1st)	12	12 ³	50	8	1
140 (2nd)	12		58	0	1
210 (1st)	23	12	0	0	2
210 (2nd)	20		5	0	2
210 (1st)	27	11	52	11	3
210 (2nd)	32		44	6	3
210 (1st)	32	13	77	55	4
210 (2nd)	31		86	33	4
210 (1st)	19	23	79	58	5
210 (2nd)	21		90	24	5
210 (1st)	31	14	25	8	6
210 (2nd)	27		33	0	6
210 (1st)	13	16	58	0	7
210 (2nd)	12		50	7	7

¹ (1st) and (2nd) = first and second spray application.

² (1) Pearce, unpublished data; (2) Busby et al. 1987a; (3) Fudge et al. 1988; (4) Busby and Pearce 1988; (5) Fudge et al. 1987; (6) Strong and Wells 1987; (7) Busby et al. 1983a. Results from sources (5) and (6) recalculated separately for 1st and 2nd spray from data in appendices.

³ Within each study, one control data set is used for comparisons with 1st and 2nd sprays.

spraying against hemlock looper is somewhat later (July) and overlaps more with the later part of the songbird breeding season when young are being reared. A relatively recent approach to forest insect control has been spraying against spruce budmoth adults in northern New Brunswick. There, spraying is in July, when most songbirds will have fledged. That may be preferable from a bird's standpoint, since fledged birds may cope better than nestlings with the stress from exposure to chemical insecticides (Grue and Hunter 1984).

Figure 3.6 gives a generalized overview of the theoretical effects of a ChE-inhibiting insecticide on songbird reproduction. As illustrated, the breeding season consists of several discrete stages. Disruptions during any single phase may result in reproductive failure although those in the early phases (territory establishment, nest site selection, nest building) are unlikely to result in complete breeding failure as re-nesting is normal in songbirds. Disruptions later in the breeding cycle, however, are more likely to result in abandonment of the breeding attempt, especially in species which are single brooded such as most Canadian songbirds.

The sublethal effects of aerial spraying of forest insecticides have been little investigated. Even when there is little influence on adult survival, reproductive success may be reduced, especially in cases when other environmental stresses (e.g. cold, wet weather following the spray) are pronounced. Reproduction may be affected by direct toxicity to the embryo, nestling or juvenile, by adult mortality, or by alteration of an adult's ability to care for the young.

Behavioural responses to OP exposure may affect free-living birds' ability to breed successfully. In laboratory studies investigators examined how exposure to OPs may affect the behaviour of songbirds and ultimately their survival or their ability to care for their young. Grue and Shipley (1981) quantified the behavioural changes in male European Starlings following the administration of a single oral dose of the OP dicotophos sufficient to cause approximately 50% brain ChE inhibition. Activities such as perching, flying, foraging, singing, and displaying were significantly reduced after treatment but returned to normal within 20-28 h. In other studies, similar responses following exposure to OPs in captive (Hill 1971; Pope and Ward 1972; Holmes 1988) and wild birds (Finley 1965; Pearce 1971; Fowle 1972) were found.

Experimental data suggest that brain ChE depression may or may not influence the reproduction of passerines by affecting the quality of parental

care given nestlings. Altricial songbirds would be expected to be particularly sensitive to changes in parental care because nestlings are dependent on their parents for both food and thermoregulation. Grue et al. (1982b) examined the effect of a single oral dose (2.5 mg/kg body weight) of dicotophos on parental care by female wild European Starlings. Dosed females made significantly fewer trips to feed their young and remained away from the nest longer than control birds. Mean brain ChE depression was 50.7% compared to controls and nestlings lost significantly more weight than control nestlings. Thus, severe but sublethal exposure may significantly reduce the quality of parental care. Reduced parental care may potentially cause a reduction in nestling growth as well as survival due to starvation, reduced fat reserves, or reduced brooding (Grue et al. 1982b). The energy intake of the nestlings may also be significantly reduced by direct exposure of nestlings to OPs through contaminated food. Alterations in the begging behaviour of nestlings, which stimulates the foraging activity of parents (von Haartman 1953), can also reduce the energy intake of nestlings.

Holmes (1988), on the other hand, reported that fenitrothion-induced ChE inhibition in excess of 50% had no demonstrable effect on reproductive success in captive Zebra Finches. However, sample sizes were limiting and the birds were exposed early in the breeding cycle (upon first introduction of male and female together in breeding cages). Consequently, because recovery of ChE inhibition to 80% of normal was projected to have occurred in 4.0-6.5 d, depending on dose, it is not surprising that subsequent effects on reproduction were absent.

In a recent study, Stromborg et al. (1988) found that the pronounced effect of dosing free-living 10-d old nestling European Starlings with a ChE-inhibiting insecticide passed within 2 d even though ChE inhibition was 19-59% (mean 46%) prior to fledging. Nestlings that survived acute exposure fledged without subsequent observable effects although 18.5% of the nestlings dosed died. Also, the authors cautioned that their results did not account for sprays where exposure may be more prolonged, where effects on adults may compound the effects on nestlings, where sprays may reduce food availability, or where exposure may occur at more critical times of the nestling period.

3.8.2.1 Territorial behaviour

The impact of fenitrothion on the territoriality of songbirds was examined in only two field studies (Lehoux et al. 1982; Busby et al. in prep.). The response of marked individuals on territories to

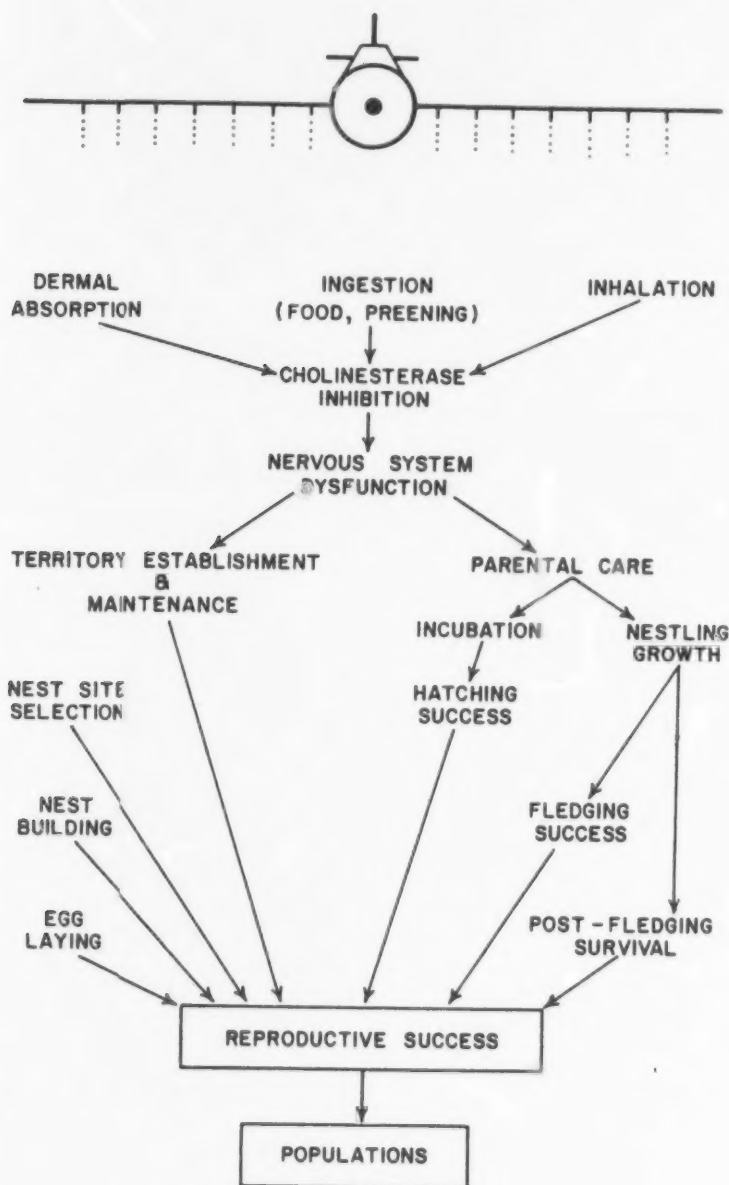


Figure 3.6 Theoretical routes of insecticide exposure in songbirds and stages of reproduction potentially affected through ChE inhibition in adults and young.

aerial application of fenitrothion was investigated in both. Effects included death, absence (death presumed), survival but desertion of territory, and replacement.

Lehoux et al. (1982) examined the fate of territorial songbirds in Quebec after an application of 1400 g a.i./ha of fenitrothion by Cessna aircraft with rotary atomizers in an evening spray. On two plots of 4 ha each and one plot of 13.5 ha, the post-spray fate of colour-banded and colour-marked individuals was examined. Although 183 birds were marked before the spray, the analysis included only those birds recaptured or seen at least twice during the pre-spray period for a total of three contacts. In all, 38 birds representing 16 species were available for analysis. The short term (3 or 4 d) impact of fenitrothion spray consisted of the disappearance of a maximum of 25% of the marked birds. The total impact was calculated as disappearance of a maximum 39% of the marked birds.

Two males, a Cape May and a Tennessee Warbler, abandoned their territories within 2 d post-spray and were replaced by individuals of the same species. Birds from all strata of the forest were affected and individuals of about ten species behaved abnormally. Extensive behavioural observations were made on two individual birds pre-spray and post-spray. One was a Cape May Warbler replaced on its territory by another individual; the other bird, a Blackburnian Warbler, exhibited a decrease in activity. Only one bird from the shrub layer was found dead, and two from the ground level were found "dead or sick."

Of the seven investigational techniques they employed, Lehoux et al. (1982) recommended monitoring of bird responses to insecticide applications by a territory mapping method. They did not, however, test the use of ChE determinations in impact assessment.

For a number of years, the impact of aerial spraying of fenitrothion has been studied in New Brunswick using White-throated Sparrows, with the assumption that birds in clear-cut areas would be as exposed to sprays as upper-canopy birds (e.g. Busby et al. 1983a). Qualitative evidence from a study by DeWeese et al. (1979) supports that assumption; Lincoln's Sparrows (*Melospiza lincolni*), which are found exclusively in open areas, were exposed to spray at almost the same frequency (82%) as were birds of species found in the upper canopy (80%).

An experimental spray of 420 g a.i./ha fenitrothion in a forest clear-cut area conducted in 1979 in New Brunswick simulated the impact which would occur

during spray overswathing or other situations resulting in greater than average exposure (Busby et al. 1983a). A second spray of 210 g a.i./ha occurred 8 d after the first spray. The 350-ha spray plot was divided equally. In one half, White-throated Sparrows were collected for brain ChE determinations and in the other half, data were collected on the territoriality and reproductive success of White-throated Sparrows (reproductive success is reported in a subsequent section).

Table 3.14 lists the fate and territoriality of 32 individually identified White-throated Sparrows on their territories. Most responses to the 420 g a.i./ha spray were immediate, taking place within the first 2 d post-spray. Approximately 19% of the marked birds were dead or missing and presumed dead after the first spray application of 420 g a.i./ha. One White-throated Sparrow (bird F10-m in Table 3.14) was found dead. One breeding pair and a female survived but deserted their respective territories in response to the 420 g a.i./ha spray. Following the first spray of 420 g a.i./ha, another individual replaced one male on its territory. Only one bird exhibited a response following the second spray of 210 g a.i./ha - a male deserted its territory the day following the spray. The total impact (including missing and dead birds, and those which deserted or were replaced on their territories) following the first spray of 420 g a.i./ha was a 31% reduction in the number of birds (Table 3.14). The impact of both sprays was a 34% reduction in the number of breeding White-throated Sparrows. In the adjacent part of the spray plot, 25% of the White-throated Sparrows collected had potentially life-threatening (i.e. $\geq 50\%$) ChE depression.

In contrast to the above study, Powell (1984) presented data indicating that Red-winged Blackbirds (*Agelaius phoeniceus*) from plots where mean brain ChE activity in collected birds was depressed 52% continued to maintain territories, and their spatial organization remained unchanged. Such study-specific differences are difficult to explain. However, species and habitat differences, as well as specific sampling protocols, may influence study results. Powell did not indicate the sampling design used except to say that ChE was "depressed 52%, 6-8 h post-treatment, and remained at that level for at least five days."

3.8.2.2 Reproductive success

Field data on the reproductive success of birds in response to fenitrothion spraying are limited. However, because of the common biochemical mode of action, information from investigations of other

Table 3.14 The fate and territoriality of White-throated Sparrows within 2 d postspray (unless otherwise noted) after application of 420 g a.i./ha fenitrothion and a second application 8 d later of 210 g a.i./ha.

<i>Bird</i> ¹	<i>Fate</i>	<i>Territory</i>
F3-m F3-f	survived survived	deserted ² remained
F5-m F5-f	survived survived	remained remained
F8-m F8-f	survived survived	remained remained
F9-m F9-f	survived survived	remained remained
F10-m F10-f	dead ³ missing	— —
F11-m F11-f	missing missing	— —
F12-m F12-f	survived survived	remained remained
F14-m F14-f	missing ⁴ missing ⁴	— —
F22-m F22-f	survived survived	remained remained
F23-m F23-f	survived survived	deserted deserted
F24-m F24-f	survived survived	remained remained
F31-m F31-f	survived survived	overtaken ⁵ remained
F32-m	survived	remained ⁶
F34-m F34-f F35-f F38-m F38-f	survived survived survived survived survived	remained remained remained ⁷ remained deserted ⁸
F39-m F39-f	survived survived	remained remained

- ¹ Nest number and sex of bird.
² Deserted territory after second spray (210 g a.i./ha).
³ Missing after first spray and found dead 7 d after spray.
⁴ Seen to be sick after first spray.
⁵ Lost territory after first spray to unmarked bird.
⁶ Lone male.
⁷ Male abandoned territory before first spray.
⁸ Deserted territory 2 d before second spray.

ChE-inhibiting insecticides becomes relevant. Evidence has been derived directly from nesting studies and indirectly from netting studies. Measurement of growth and development in nestling altricial birds offers interesting possibilities for assessment of sublethal effects of insecticides.

3.8.2.2.1 Nesting studies

In Laughing Gulls (*Larus atricilla*), exposure to OPs in the field has been implicated in nestling mortality which reduced fledging success by as much as 25% (White et al. 1979) and 33% (White et al. 1983a). Mortality of nestlings probably resulted from both direct exposure through contaminated food and from starvation resulting from reduced nest attentiveness by adults. A single dose of parathion (6 mg/kg) given to one member of the adult pair was found to affect incubation behaviour (White et al. 1983b) but a single dose of 5 mg/kg did not affect nest defense behaviour, as defined by flushing distance upon being approached by a human, or reproductive success (King et al. 1984). In the latter study, the authors cautioned that single sublethal doses to only one adult may not adequately simulate actual field exposure as found in the study by White et al. (1979, 1983a).

Powell (1984) studied the effect of aerial application of fenthion (an OP insecticide highly toxic to birds, Hudson et al. 1984), used in mosquito control, on reproduction in an altricial species - the Red-winged Blackbird. Depression of brain ChE activity did not affect the birds' ability to maintain a territory nor did it result in nest desertion. Hatching and fledging success in birds exposed similarly (i.e. birds were assumed to have brain ChE activity inhibited by about 52%, based on known ground deposit) were not affected. Although significantly lower during the first 3 or 4 d after hatching, overall growth rate of nestlings was not affected. It is interesting that the insecticide did not affect nestling brain ChE activity and that fenthion residues were not detected in food fed to the nestlings even though residues were found in sweep net samples of arthropods collected after the treatment. Perhaps either contaminated prey became unavailable to foraging birds or the adults were able to detect and avoid such prey.

The effect of fenitrothion spraying on the reproductive success of altricial bird species was investigated in two field studies. In reviewing those studies, the following definitions of reproductive parameters are used. Hatching success is the number of eggs hatched divided by the number laid. Fledging success is the number of young fledged divided by the number of

eggs hatched. Overall reproductive success is the number of young fledged divided by the number of eggs laid. All are expressed as a percentage.

The reproductive success of White-throated Sparrows was examined in a cut-over area sprayed with 420 g a.i./ha fenitrothion and sprayed again at 210 g a.i./ha 8 d later, and in a comparable control area (Busby in prep.). The data were examined by analysis of variance of arcsine-transformed reproductive parameters for each nest. Hatching success was lower in the sprayed area although the difference was not statistically significant. In the spray plot, both fledging success and overall reproductive success were lower ($P < 0.05$) than in the control area (Table 3.15). Overall reproductive success in the spray plot was less than one-third that in the control plot. In the spray area, after the first spray, 95% of the White-throated Sparrows sampled had 20% or more brain ChE depression indicating a significant exposure of almost all birds to the spray. Twenty-five percent had post-spray ChE depression 50% or more (Busby et al. 1983a). The mean depression for all days was 42%, a maximum mean depression of 52% occurring on the day of spray. In White-throated Sparrows, therefore, significant reproductive failure was associated with 40-50% brain ChE depression.

Parents at five of the nests in the sprayed plot, but none in the control plot, were "desultory" incubators; their incubation became irregular following the application of 420 g a.i./ha, eggs sometimes becoming cold to the touch. Although eggs did eventually hatch, incubation was prolonged to 13-17 d from a normal incubation period of 11-14 d. Therefore some disruption during incubation may not necessarily cause reproductive failure. However, abandonment of the nest for only a day, once the young have hatched, would be likely to result in nestling mortality. Thus, fledging success may be a more sensitive measure of the impact on reproduction of fenitrothion sprays than hatching success or overall reproductive success.

In Scotland, where single sprays of fenitrothion were applied by helicopter at a dosage rate of 300 g a.i./ha using ULV techniques, Spray et al. (1987) reported that the insecticide did not affect the proportion of broods hatched or fledged, clutch size, nestling growth, brood size at hatching, or fledging of Coal Tits breeding in nest boxes. Because nestlings were protected from direct exposure, reduced reproductive success would likely be the result of altered parental behaviour or contaminated food and not direct exposure to the spray. Spray et al. (1987) also reported reproductive success in terms of

nest success - a method in which only one nestling need survive to fledging for a nest to be considered 100% successful, potentially masking any impact of the sprays on reproductive success. Although the authors conclude there was no effect of the spray on reproduction, details regarding the relative timing of spraying and breeding phenology were not provided. Thus it is difficult to evaluate the degree of exposure experienced by the birds.

3.8.2.2.2 Netting studies

Mist-net studies of captures of birds to examine the production of young (i.e. fledging success) following fenitrothion spraying were reported by Lehoux et al. (1982) and Millikin (1987). Some of the biases associated with mist netting have already been discussed in detail (see Avian Impact Assessment Techniques).

Lehoux et al. (1982) found that the proportion of juveniles of all species captured with mist nets did not differ significantly between control and sprayed areas after application of fenitrothion at 1400 g a.i./ha. Unfortunately, only a few individuals of any particular species were captured and, therefore, data analysis was made without the benefit of species distinctions and without reference to species-specific dispersal patterns of juveniles. The authors pointed out that netting did not provide a representative sampling of the bird community because canopy species were under-represented and species of the shrub layer were over-represented. While the lack of

canopy species in their samples was probably a netting bias, the possibility of it also being an effect of the insecticide spraying cannot be ruled out. In addition, Lehoux et al. (1982) reported that the young of most species in their spray plots had not hatched by spray application time and thus the impact on nestlings would be minimal except as a result of alteration in incubation behaviour of the parents. As the hatching peak occurred 10 d after spray, the exposure of nestlings to contaminated food and the lack of food due to the incapacitation of parents would be minimal.

Millikin (1987) also used mist netting captures to examine the production of young on sprayed (280 g a.i./ha fenitrothion applied by Cessna aircraft) and unsprayed plots. Fledging success was reduced in the sprayed area during the year of treatment when compared to values for the same plot during non-sprayed years and to control plots. The total (i.e. all species) proportion of fledglings and adult males captured was lower than that reported by Lehoux et al. (1982) even though the dosage rate of fenitrothion was substantially lower (280 vs. 1400 g a.i./ha). Millikin did not report absolute netting effort (i.e. the total number of netting hours), nor sample sizes of birds caught. In addition, she gave no justification as to why warblers were reported as a group while other species (Purple Finch (*Carpodacus purpureus*), White-throated Sparrow), were reported singly. Neither did Millikin discuss how representative of the bird community her net captures were nor when the peak hatch date occurred in relation to spray date.

Table 3.15 Hatching, fledging, and overall reproductive success of White-throated Sparrows in control and sprayed (fenitrothion at 420 g a.i./ha + 210 g a.i./ha 8 d later) areas.

Mean Reproductive Parameters ¹	Control (N=7 nests, 28 eggs)	Sprayed (N=13 nests, 48 eggs)
% Hatching Success	71	58(ns)
% Fledging Success	92	29*
% Reproductive Success	64	18*

¹ Hatching success = (no. eggs hatched/no. eggs laid) X 100.

Fledging success = (no. birds fledged/no. eggs hatched) X 100.

Reproductive success = (no. birds fledged/no. eggs laid) X 100.

ns No significant difference between Controls and Sprayed, ANOVA.

* Sprayed significantly lower than Controls (P<0.05), ANOVA.

Moulding (1976) suggested that insecticide-induced nesting failure with concurrent food depression may result in birds nesting in other areas the following year. Because Millikin's study took place over a number of years, she was able to calculate return rates of birds among years. In the spray plot, the return rate of individuals in the year following treatment was significantly lower than that in the control plot in any year, as well as in previous years in the treatment plot when no sprays took place. She could not, however, distinguish whether the lower return rate of birds was caused by increased mortality or due to nesting in other areas.

3.8.2.2.3 Nestling growth

Several authors (e.g. Jackson and Park 1973; Powell and Gray 1980; Grue et al. 1982b; Grue and Shipley 1984) referred to reductions in mass in altricial nestlings following exposure to ChE-inhibiting insecticides. However, only one study of the effects of fenitrothion on growth of free-living songbirds has been published, Spray et al. (1987) examining the effects of fenitrothion spray on the growth of nestling Coal Tits. They found no statistically significant differences in growth, as measured by weight, tarsus and wing length, among nestlings in control and in sprayed areas. However, they noted that their data indicated that nestling increase in weight was slower, which resulted in a lower final weight of nestlings in sprayed areas, and which suggested that the technique for measuring growth used may not have been sensitive enough. It is difficult to evaluate their growth analysis independently because of limited information on the method used. Zach (1988) provided a critical evaluation of growth-curve analysis and the numerous methodological difficulties involved.

The Canadian Wildlife Service investigated fenitrothion effects on White-throated Sparrow nestling growth and development in free-living birds. Separate studies addressed spray events, oral dosing of nestlings, and oral dosing of adult birds. Findings of those studies were reported (Peters 1979; Pearce and Busby 1980; Busby 1982b; Busby and Pearce 1984) but remain unpublished. There were two main difficulties in analyzing the results: a high degree of inter- and intra-nest variability shown in the data, even in control areas, made it impossible to isolate insecticide-induced effects; and finding sufficient nests to provide the large data sets necessary for statistical confidence was logistically prohibitive. Nevertheless, simple plots of growth statistics comparing treated birds with controls indicated that nestlings, particularly young ones, are sensitive to fenitrothion. If pursued, measurement of growth

may be a viable impact assessment technique. Zach (1988) suggested a more simplified approach to nestling growth assessment.

3.8.2.3 Effects of insecticide-induced food depression

Several authors suggested that birds shift their foraging activity from sprayed areas to nearby unsprayed ones in response to insecticide-induced food depression (e.g. Moulding 1976; Richmond et al. 1979; DeWeese et al. 1979). That hypothesis was examined in the field in only two studies.

Hunter and Witham (1985) examined arthropod abundance and the foraging behaviour response of warblers to aerial application of carbaryl (a ChE-inhibiting carbamate) in Maine spruce forests. There was a reduction in the number of warblers foraging in the sprayed area. Birds foraged lower in the canopy and in a greater variety of trees for 7 d following spraying in 1980. In the following year, no change in numbers or foraging behaviour followed spraying. Hunter and Witham (1985) suggested that the spray coverage was patchy in 1980 and birds responded by moving to nearby unsprayed foraging locations within the spray block. In 1981, uniform spray coverage prevented a similar response by warblers because nearby unsprayed areas were not available.

Millikin (1987) examined the foraging location in trees by three species of birds following spraying of 280 g a.i./ha of fenitrothion. Chestnut-sided Warblers (*D. pensylvanica*), which forage in deciduous trees, did not change their foraging activity in response to spraying. Magnolia Warblers reduced their foraging activity in conifers. White-throated Sparrows changed their foraging to areas which hypothetically received less spray deposit.

Powell (1984) found that the insecticide fenitrothion did not affect the spatial organization and territoriality of Red-winged Blackbirds even though it reduced the insect biomass significantly. He suggested that in years of high food abundance, such as in the one in which the investigation was done, a reduction in food supply may not hinder the ability of birds to complete nesting successfully whereas in years of limited food availability the loss may be more critical.

The implications of shifts in foraging locations by birds in response to the pattern and extent of spray deposit are not known. Travel time and energetic costs of foraging would be expected to increase when birds are forced to forage off their territories. Millikin (1987) suggested that reproductive success may be

reduced when large tracts of forest are sprayed with fenitrothion at operational dosage rates but reproductive success is unaffected when nearby unsprayed areas are available for foraging. It is possible that the reduced reproductive success (as measured by mist netting) observed in Millikin's study was not the result of food depression but of exposure to the insecticide of both parents and young.

3.8.3 Acute Responses in Wild Birds

Estimates of mortality in relation to fenitrothion spraying operations have relied primarily on detection of bird carcasses, marked reduction in singing activity as measured in bird censuses, and observations of abnormal behaviour indicative of acute poisoning.

Buckner (1974) and NRCC (1975) gave the following summarization based on then currently available data regarding the range of acute responses of forest birds in relation to dosage rate of fenitrothion. There were reports of behavioural abnormalities and reduced singing and movements in canopy dwelling species at dosage rates as low as 138 g a.i./ha. At rates of 206-275 g a.i./ha, adults of canopy dwelling species behaved abnormally and there were reports of some juvenile mortality. Mortality occurred in adults inhabiting the crown canopy at dosage rates above 275 g a.i./ha. At rates of 550 g a.i./ha and greater, adult mortality increased markedly and was no longer limited to species inhabiting the canopy.

Buckner (1974) reported a range of acute effects at dosage rates of 1238 g a.i./ha of fenitrothion. In some species, there was complete mortality; in others, there were substantial reductions in numbers. Some species experienced impaired fledging, and yet other species appeared to be completely unaffected.

Buckner (1974) presented mortality curves in relation to dosage rates of fenitrothion for three species of birds. No information was given on how the curves were calculated, and our efforts to obtain details were unsuccessful. The data apparently were collected in several different studies of various operational dosage rates as well as accidental overdoses (S. B. Holmes, Canadian Forestry Service, Sault Ste. Marie, pers. comm.). The curve is relatively steep for Chipping Sparrow (*Spizella passerina*) and Common Yellowthroat (*Geothlypis trichas*), particularly for juveniles. Adult and juvenile White-throated Sparrows appear to be relatively unaffected even when the dosage rate was 1400 g a.i./ha. Buckner's graph suggests that the tolerance of forest

songbirds to fenitrothion is relatively narrow and that even small increases in the dosage rate could result in a sharply increased level of impact (NRCC 1975).

Bart and Hunter (1978) made a detailed analysis of studies in which searches for sick and dead birds were conducted. They indicate that the dosage rate at which casualties begin to be found appears to be 280 g a.i./ha. However, Bart and Hunter (1978) found that results from singing-bird surveys were not as consistent as those from carcass searches. In some studies, decreased singing was noted in areas treated with 280 g a.i./ha fenitrothion while in other studies it was found that canopy birds persisted (i.e. were heard singing) in areas treated with up to 560 g a.i./ha. Variability in the results of singing-bird censuses is expected in view of the numerous factors, including the high variability in spray deposit, that can bias the results.

Lehoux et al. (1982), reporting on observations of an extremely high dosage (1400 g a.i./ha) spray of fenitrothion, gave conflicting results with regard to mortality within the spray block. Table 29 of that report indicates that only three sick birds were found, whereas Table 30 reports one dead bird and two "dead or sick" birds found. In any case, their findings suggest a very low rate of mortality, based on searching over a period of 12 d post-spray. As several authors noted (see sub-section on Carcass Searches), dead birds are usually found within 2 d post-spray. Thus Lehoux's search effort in the critical period may have been inadequate, resulting in an underestimation of mortality based on carcass searches.

The finding of sick, moribund, and dead birds at operational dosage rates appears not to be predictable. Canadian Wildlife Service workers conducting singing-bird censuses consistently found sick, moribund, and dead birds after fenitrothion spraying. Data in Table 3.16 (Pearce 1967, 1968, Canadian Wildlife Service) indicate some spray casualties found in every year from 1967 to 1976 (except 1969 when fenitrothion studies were not conducted). The table does not include casualties found in situations where other chemicals could have been causative factors (e.g. if phosphamidon was used in the first spray, casualties found after a subsequent fenitrothion spray were excluded). Dead birds were also found following fenitrothion spraying in Maine (Peterson 1969, 1971, Table 3.16). As indicated in Table 3.16, spray casualties have been found at the lowest operational dosage rate (140 g a.i./ha) as well as at the higher ones. In contrast, after reviewing numerous reports of studies of

fenitrothion effects on birds by the Forest Pest Management Institute (formerly part of the Chemical Control Research Institute) of the Canadian Forestry Service, no specific reference to spray casualties could be found. It is interesting to note that Canadian Forestry Service researchers generally used a different assessment method (small plot censuses) from the one used by Canadian Wildlife Service investigators (transects) to assess impacts of insecticides on birds and, as discussed below, therein may lie the reason they have not found spray casualties.

Pearce et al. (1979a) found further evidence for unpredictable results in searches for spray casualties. Spray casualties, although common, were not found in all fenitrothion sprayed areas or on all transect

routes in a given spray regime although the searching method and effort remained similar.

Since the Pearce et al. (1979a) study, which took place in 1976, avian mortalities in relation to operational spraying of fenitrothion were found in only one investigation. In 1987, two dead birds, a Pine Siskin and a Yellow-rumped Warbler were found in a spray area which received 210 g a.i./ha fenitrothion using UULV spray techniques (Busby and Pearce 1988). In the same study, one Black-throated Green Warbler (*D. virens*) was observed to be incapacitated (reluctance to fly, short erratic flights, sought dense cover in shrubs and on ground) but not captured. Both dead birds had very low brain ChE activity, as discussed earlier (Toxicology: Acute and Subacute Responses).

Table 3.16 Birds found sick, moribund, or dead in spray blocks after fenitrothion spraying of forests in New Brunswick, Newfoundland and Maine, 1967 - 1987¹.

Year	Dosage rate (g a.i./ha)	No. birds found			Total	Source
		Sick	Moribund	Dead		
1967	560	13	2	1	16	Pearce (1967)
1968	560	7	0	0	7	Pearce (1968)
	420	15	0	0	15	
	280	6	0	2	8	
	420	0	0	4	4	Peterson (1969)
1970	140	0	0	1	1	Peterson (1971)
	140	0	0	6	6	
1971	140	0	0	4	4	Canadian Wildlife Service, unpublished data
1972	210	3	0	0	3	"
1973	210	0	0	4	4	"
1974	210	0	0	11	11	"
	175	0	0	3	3	
1975	175	5	0	14	19	"
1976	210	3	4	11	18	"
1979	420	0	3	1	4	"
1987	210	0	0	2	2	"

¹ This table does not include the approximately-similar numbers of birds which were found sick, moribund, or dead but which could not with complete certainty be associated with a specific fenitrothion spray block, even though subsequent analyses revealed fenitrothion residues in them. Nor does it include the numerous "sightings" of sick birds which were not captured (e.g. Pearce 1968).

The near absence of spray casualties after 1976 probably reflects different directions taken by researchers rather than an elimination of the problem. The Canadian Wildlife Service did not conduct fenitrothion studies in 1977, and thereafter that agency turned to studies of effects on reproduction, brain ChE assessments, and investigations of other insecticides.

From the information given above, it is concluded that if searching for spray casualties is to be used as a quantitative assessment technique, systematic searches should be conducted in a number of areas subjected to the same spray regime to obtain a better estimate of insecticide-induced responses.

Many other studies reported no dead birds found and no birds behaving abnormally in fenitrothion-sprayed forest (e.g. Buckner and McLeod 1977; Busby et al. 1981, 1987a; Kingsbury and McLeod 1981; Fudge et al. 1987, 1988; Strong and Wells 1987). There are reasons why those investigations cannot be used as evidence that avian mortality resulting from fenitrothion spraying does not occur. Firstly, the primary focus of those studies was not on carcass searches. The probability of negative findings from casual searches for carcasses is very high, as discussed earlier. Secondly, in earlier carcass searches a much greater area was surveyed and more variable spray conditions were encountered. Pearce et al. (1976, 1979a) conducted counts on transects two to three or five km long, and many were conducted yearly. Finally, studies by Pearce et al. (1976, 1979a) clearly indicate that avian mortality will not be detected following every spray event despite a consistent search effort. That may be because mortality does not occur following every spray event or because the probability of detection is generally very low, or both. In the study by Pearce et al. (1979a), dead birds were found on up to one-half of the transects surveyed, depending on the spray regime. Thus, if the probability of avian mortality is lower than 100%, spray regimes were not replicated, study plots were small, and only casual searches were made, it is not surprising that in many recent studies dead birds were not found in fenitrothion spray zones.

In the past, there were reports from the public of avian mortality in sprayed areas (Pearce 1967, 1968; Peterson 1976). Such reports have become much rarer in recent years. Since 1977, regulatory buffer zones around human habitations, municipal water sources, lakes and rivers, and other ecologically sensitive areas (such as blueberry fields) have come into effect where spraying operations are prohibited

(Sexsmith 1987). Pearce et al. (1976) noted that most public reports of avian mortality came from highly populated areas. Observations of avian mortality would now be less likely to be made by the public because spraying operations no longer take place near human habitation.

Fenitrothion is registered for forest use in Canada at dosage rates ranging from 140 to 280 g a.i./ha, with a minimum interval between two sprays of 5 d. It is clear that some adult bird mortality will occur in the course of operational fenitrothion spraying under present regulations. Unfortunately, there is currently no way to assess the magnitude or frequency of that mortality. Given the difficulty of finding carcasses, especially in a forest setting, and given also that mortality occurred at the lowest dosage rate currently registered, it is likely that adult songbird mortality is far more extensive than documented. It is also reasonable to assume that nestling and juvenile mortality occur before adult survival is affected.

As stated earlier, knowledge of the nominal dosage rate of fenitrothion is inadequate in predicting the risk to forest birds. The total amount of spray deposit and deposit pattern in relation to dosage rate appear to be unpredictable. Many factors associated with spray application could lead to incidences of over-exposing birds and may include navigational errors, tank mixing errors, lack of contractor compliance with regulations, poor equipment calibration, forest structure, and meteorological and topographic factors causing higher than average deposits. The lethal threshold for birds of a given dosage rate may thus vary between different spray regimes or even within the same spray regime because of the wide-ranging level of deposits associated with aerial application of forest insecticides.

Various factors may result in forest birds being more susceptible to forest spraying, or those factors may potentiate its effects. The diurnal activity patterns of birds in relation to the timing of sprays (i.e. morning vs. evening sprays), poor physiological condition, and weather-related stresses, such as low temperatures, are all factors tending to increase the susceptibility of birds to a particular spray event.

3.8.4 Insecticide Residue Analysis

Because enzyme inhibition is a cumulative process and metabolism causes the rapid breakdown and excretion of the compounds causing inhibition (NRCC 1975), it is difficult to quantitatively relate ChE activity, or behavioural changes, to fenitrothion residues.

During the years 1970-1976, Canadian Wildlife Service personnel conducting singing-bird censuses in fenitrothion spray blocks found sick, moribund, or dead songbirds considered to be spray victims. All specimens for which other possible causes of death appeared to be extremely unlikely were sent to the laboratories of the Ontario Research Foundation for residue analyses. The carcasses were stored frozen (-20°C) pending analysis for residues of fenitrothion during the following winter. Analyses were on individual birds from which the head, feet, remiges, and rectrices had been removed. Some topical residues may have been removed while the carcasses were being thus prepared. The analytical method involved Soxhlet extraction, reverse-phase column clean-up, and final analysis by GLC using the flame photometric detector.

The results of those analyses are presented in Table 3.17 and indicate an extremely high degree of variability among individuals and spray dosage rates. Some of the highest residues were found in casualties from forest treated at the lower spray dosage rates (140 and 175 g a.i./ha) whereas some of the lowest were associated with the highest spray dosage rate (210 g a.i./ha). Some birds found dead or moribund had no detectable fenitrothion residues (i.e. <0.01 ppm) whereas Busby et al. (1983a) collected a live, apparently well, White-throated Sparrow containing 7.0 ppm residual fenitrothion.

Busby et al. (1983a) found a significant relationship between ChE activity and fenitrothion residues at a spray dosage rate of 420 g a.i./ha. Data presented by Hamilton et al. (1981) generally indicate that higher whole body residues are associated with lower ChE activities. Thus, although measurement of tissue residues appears to be a relatively poor method for determining the impact of fenitrothion on wild birds, further research may identify a greater usefulness of that approach.

3.9 POPULATION EFFECTS

Three main difficulties are involved in assessing the impact of insecticide use at the species population level. The first involves defining the boundaries of the population of a species of interest. For those species with restricted numbers or geographical ranges, it is possible to measure the size of the entire population. In practice, populations are usually defined only for endangered and exploited species. Researchers usually define population boundaries with reference to their own particular study areas.

The second difficulty of examining effects of insecticides at the population level is distinguishing between pesticide effects and those of other factors such as habitat change and adverse weather. The necessary field studies would be labour intensive and involve long-term commitments. The third problem associated with population studies is that of scale. Most investigations which have addressed population effects of fenitrothion were restricted to plots smaller than recommended for census methods, ranging from 4 to 8 ha (e.g. Buckner and McLeod 1977; Kingsbury and McLeod 1981; Millikin 1987). Census techniques which in themselves have inherent methodological problems were invariably used. Small-scale studies may only provide insight into the processes that influence population size when the movement of individuals into and out of the study area is small compared to replacement of individuals by reproduction within the restricted study area (Ricklefs 1979), a condition which is usually not met.

Pearce et al. (1979a) examined Breeding Bird Survey data for ten species of forest birds in New Brunswick for 1967-1976. Indices to changes showed no downward trends during that decade of fenitrothion use although the authors recognized problems relating to the inconsistent inclusion of survey routes, or parts thereof, in spray zones.

Population effects should ideally be examined on a scale relevant to the population and with reference to the extent of the spray program. That becomes particularly important when assessing the impact of extremely large forest spray programs such as those in eastern Canada (e.g. New Brunswick 1976 spray program, Figure 3.1) where areas sprayed may encompass significant portions of a species' or subspecies' range. However, definition of boundaries between populations or subpopulations is poorly understood for most North American songbird species and examination of songbird populations for large areas is logistically prohibitive.

Detailed data on the population status of songbirds in eastern Canada are currently not available. As Mineau and Peakall (1987) suggested, the only feasible method for determining population effects of pesticides is to examine the influence on reproduction, the assumption being that if reproduction is impaired, concern for population integrity is warranted. Population responses to pesticides could best be examined through mathematical modelling addressing reproductive effects, the extent and intensity of spray operations, and other relevant factors.

Table 3.17 Fenitrothion residues (ppm wet weight) in songbirds found sick (S), moribund (M), or dead (D) following fenitrothion sprays in New Brunswick, 1970-1976 (Canadian Wildlife Service, unpublished data).

Year	Sub-family	N	Mean/ range	Spray dosage rate	Species and condition when found ¹
1970	Parulinae	6	2.14 0.16, 5.22	140 ²	BLWA, 2 (D,D); CMWA, 1 (D); AMRE, 3 (D,D,D).
1971	Parulinae	3	2.39 1.62, 3.48	140	YRWA, 1 (M); MAWA, 2 (M,M).
	Carduelinae	1	2.56	140	PUFI, 1 (M).
1972	Parulinae	2	1.06 0.14, 1.98	210	BTGW, 1 (S); MAWA, 1 (S).
	Emberizinae	1	4.76	210	DEJU, 1 (S)
1973	Fluvicolinae	1	ND ³	210	OSFL, 1 (D/M ⁴).
	Parulinae	1	ND	210	CSWA, 1 (D/M).
	Cardinalinae	2	0.05 ND, 0.10	210	RBGR, 2 (D/M).
1974	Sylvinae	1	0.69	210	RCKI, 1 (D/M).
	Parulinae	8	0.65 ND, 2.16	210	BBWA, 2 (D/M,D/M); AMRE, 1 (D/M); BTGW, 1 (D/M); MAWA, 1 (D/M); YEWA, 1 (D/M); NAWA, 2 (D,D).
1975	Turdinae	1	0.04	175	SWTH, 1 (D).
	Parulinae	7	2.75 0.25, 5.97	175	WIWA, 1 (D); BPWA, 1 (D); AMRE, 1 (D); BLWA, 1 (D); CMWA, 1 (D); MAWA, 2 (D,D).
	Emberizinae	9	3.73 0.02, 7.02	175	CHSP, 4 (D,S,S,S); LISP, 1 (D); VESP, 1 (D); DEJU, 2 (D,S); WTSP, 1 (S); PISI, 1 (D).
	Carduelinae	1	10.4	175	
1976	Parulinae	17	1.08 0.13, 3.06	210	BBWA, 1 (D); AMRE, 3 (D,D,M); COYE, 1 (M); CAWA, 1 (D); NPWA, 1 (D); BTGW, 4 (D,D,M,S); BLWA, 3 (D,D,S); BPWA, 1 (S); YRWA, 2 (D,M).
	Turdinae	1	0.02	210	AMRO, 1 (D).

¹ Species abbreviation followed by sample size for that species. Species abbreviations as follows: OSFL-Olive-sided Flycatcher; RCKI-Ruby-crowned Kinglet; SWTH-Swainson's Thrush; AMRO-American Robin; Warblers-NAWA-Nashville, NPWA-Northern Parula, YEWA-Yellow, CSWA-Chestnut-sided, MAWA-Magnolia, CMWA-Cape May, YRWA-Yellow-rumped, BTGW-Black-throated Green, BLWA-Blackburnian, BBWA-Bay-breasted, BPWA-Blackpoll, AMRE-American Redstart, COYE-Common Yellowthroat, WIWA-Wilson's, CAWA-Canada; RBGR-Rose-breasted Grosbeak; Sparrows-CHSP-Chipping, VESP-Vesper, LISP-Lincoln's, WTSP- White-throated; DEJU-Dark-eyed Junco; PUFI-Purple Finch; PISI-Pine Siskin.

² Spray dosage rate expressed in g a.i./ha.

³ ND = no fenitrothion residues detected (i.e. <0.01 ppm).

⁴ D/M = Dead or moribund.

3.10 CONCLUSIONS

As this review of the avian impacts of fenitrothion use in forestry is concluded, the usual dilemma is faced: despite the considerable amount of information available, there remain more questions than answers. However, the authors would fail their statutory responsibilities as resource managers if an attempt was not made to provide an overall assessment of the situation. There are, in fact, more environmental and other data available on fenitrothion than on most agricultural insecticides put together.

On the positive side, there have not been reports of precipitous declines of bird species following long-term use of fenitrothion in New Brunswick or other jurisdictions. It is known that the impact on birds was more severe with another forestry insecticide, phosphamidon. The effects of fenitrothion use appear to be variable even at very high spray dosage rates. Some spray events may result in minimal or no impact.

On the other hand, there is evidence that factors other than dosage rate have a bearing on impact - mortality of birds having been noted at every dosage rate monitored. Those factors are inherent in forest spraying and are unlikely to be brought under complete control. Birds with brain ChE depression of more than 50% have been collected in every fenitrothion spray block monitored using that method of impact assessment. It appears that death or incapacitation may prevent the sampling of other birds with that or a greater level of ChE depression. Of the two studies in which reproduction in field-exposed birds was examined, one showed a drastic reduction in nest success associated with a mean brain ChE depression of 42% and very low visible mortality.

One of the most disturbing aspects of all the evidence gathered to date is that, as repeatedly discussed in this review, most methodological biases consistently result in underestimation of exposure and impact. That may not be a concern if fenitrothion use was occasional and small scale. The problems specific to large insect control programs, which strive to deliver a chemical to large areas within a short period of time, make risk assessment even more complex (McEwen 1981).

It is concluded that, whereas spraying of fenitrothion in the present use pattern might be acceptable on a small scale as an insecticide of last resort, there are serious doubts about the desirability of such an insecticide being in the arsenal of products available

for broadscale forest spraying. Other chemicals used operationally or experimentally for budworm control (e.g. aminocarb, mexacarbate) are less impactful on birds than fenitrothion (Pearce and Garrity 1981; Busby et al. 1982; Busby et al. 1983b; Busby et al. 1987b). Unfortunately, there are current availability problems. A powerful tool for forest protection is the microbial insecticide B.t. Now is an appropriate time to reassess the role that fenitrothion will play in forest protection and to turn collective attention away from that insecticide to more promising alternatives.

3.11 ACKNOWLEDGEMENTS

Thanks are expressed to B. Collins for advice and support in the statistical analysis of data. We are also much indebted to H. Crick, D. Forsyth, and S. Holmes for providing unpublished information.

3.12 LITERATURE CITED

- Armstrong, J.A. 1977. Relationship between the rates of pesticide applications and the quantity deposited in the forest. In J.R. Roberts, R. Greenhalgh, and W.K. Marshall (eds.). Proc. symp. on fenitrothion: The long-term effects of its use in forest ecosystems. NRCC Publ. No. NRCC 16073, Ottawa. p. 183-202.
- Armstrong, J.A. and W.N. Yule. 1978. The distribution of aerially-applied spray deposits in spruce trees. Can. Entomol. 110: 1259-1267.
- Ayer, W.C., G.L. Brun, D.C. Eidt, W. Ernst, V.N. Mallet, R.A. Matheson, and P.J. Silk. 1984. Persistence of aerially applied fenitrothion in water, soil, sediment, and balsam fir foliage. Can. For. Serv., Mar. For. Res. Centre Inf. Rep. M-X-153.
- Balcomb, R. 1986. Songbird carcasses disappear rapidly from agricultural fields. Auk 103: 817-820.
- Barry, J.W. 1984. Deposition of chemical and biological agents in conifers. In W.Y. Garner and J. Harvey, Jr. (eds.). Chemical and biological controls in forestry. Am. Chem. Soc. Symp. Ser. No. 238: 117-138.

- Barry, J.W., W.K. Ciesla, M. Tysowski, and R.B. Ekblad. 1977. Impaction of insecticide particles on western spruce budworm and Douglas fir needles. *J. Econ. Entomol.* 70: 387-388.
- Bart, J. 1979. Effects of acephate and sevin on forest birds. *J. Wildl. Manage.* 43: 544-549.
- Bart, J. and L. Hunter. 1978. Ecological impacts of forest insecticides: An annotated bibliography. Canada-U.S. Spruce Budworms Program Rep.
- Bignami, G., N. Rosic, H. Michaleck, M. Milosevic, and G.L. Gatti. 1975. Behavioural toxicity of anticholinesterase agents: methodological, neurochemical and neuropsychological aspects. In B. Weiss and V.G. Laties (eds.). *Behavioral Toxicology*. Plenum Press, New York. p. 155-211.
- Bolton, T.B. 1976. Nervous system. In P.D. Sturkie (ed.). *Avian Physiology*. Third Edition. Springer-Verlag. New York. p. 1-28.
- Breslow, N.E. and N.E. Day. 1980. Statistical methods in cancer research. Vol. 1. The analysis of case control studies. IARC Scientific Publication No. 32. World Health Organization, Lyon, France. 338 p.
- Buckner, C.H. 1974. The biological side-effects of fenitrothion in forest ecosystems. *Can. For. Serv., Chem. Control Res. Inst., Inf. Rep.* CC-X-67.
- Buckner, C.H., B.B. McLeod, and D.G.H. Ray. 1973. The effect of operational application of various insecticides on small forest birds and mammals. *Can. For. Serv., Chem. Control Res. Inst., Inf. Rep.* CC-X-43.
- Buckner, C.H. and B.B. McLeod. 1977. Impact of experimental spruce budworm (*Choristoneura fumiferana* Clemens) suppression trials upon forest-dwelling birds in Newfoundland in 1977. *Can. For. Serv., For. Pest Manage. Inst. Rep.* FPM-X-9.
- Bunyan, P.J., D.M. Jennings, and A. Taylor. 1968. Organophosphorus poisoning: diagnosis of poisoning in pheasants owing to a number of common pesticides. *J. Agric. Food Chem.* 16: 332-339.
- Busby, D.G. 1982a. Food habits of nestling White-throated Sparrows. In D.R. Townsend (ed.). *Environmental surveillance of spray operations for forest protection in New Brunswick, 1980-81*. New Brunswick Dept. Environ. p. 64-65.
- Busby, D.G. 1982b. Effects of oral dosing with fenitrothion on the growth and development of nestling White-throated Sparrows. In D.R. Townsend (ed.). *Environmental surveillance of spray operations for forest protection in New Brunswick, 1980-81*. New Brunswick Dept. Environ. p. 63.
- Busby, D.G. and P.A. Pearce. 1984. Response of songbirds to exposure to fenitrothion. In *Environmental monitoring of forest insect control operations*. New Brunswick Dept. Environ. p. 11-13.
- Busby, D.G. and P.A. Pearce. 1988. The effect of fenitrothion, applied by rotary atomizer-equipped TBM aircraft, on brain cholinesterase activity in forest songbirds. In *Environmental monitoring of forestry control operations. 1988 EMOFCO Rep.*, New Brunswick Dept. Municip. Affairs and Environ. p. 30-34.
- Busby, D.G., P.A. Pearce, and N.R. Garrity. 1981. Brain cholinesterase response in songbirds exposed to experimental fenitrothion spraying in New Brunswick, Canada. *Bull. Environ. Contam. Toxicol.* 26: 401-406.
- Busby, D.G., P.A. Pearce, and N.R. Garrity. 1982. Brain cholinesterase inhibition in forest passerines exposed to experimental aminocarb spraying. *Bull. Environ. Contam. Toxicol.* 28: 225-229.
- Busby, D.G., P.A. Pearce, and N.R. Garrity. 1983b. Brain ChE response in forest songbirds exposed to aerial spraying of aminocarb and possible influence of application methodology and insecticide formulation. *Bull. Environ. Contam. Toxicol.* 31: 125-131.
- Busby, D.G., P.A. Pearce, N.R. Garrity, and L.M. Reynolds. 1983a. Effect of an organophosphorus insecticide on brain cholinesterase activity in White-throated Sparrows exposed to aerial forest spraying. *J. Appl. Ecol.* 20: 255-263.

- Busby, D.G., P.A. Pearce, and N.R. Garrity. 1987a. Effect of ultra ULV fenitrothion spraying on brain cholinesterase activity in forest songbirds. *Bull. Environ. Contam. Toxicol.* 39: 304-311.
- Busby, D.G., S.B. Holmes, P.A. Pearce, and R.A. Fleming. 1987b. The effect of aerial application of Zectran on brain cholinesterase activity in forest songbirds. *Arch. Environ. Contam. Toxicol.* 16: 623-629.
- Chansler, J.F. and J.B. Dimond. 1971. Effects of Accothion on the spruce budworm (intensive investigations on small plots). In R.W. Nash, J.W. Peterson, and J.F. Chansler (eds.). *Environmental studies in the use of Accothion for spruce budworm control. A cooperative study by the State of Maine, U.S. Dept. Agric., and U.S. Dept. Interior.* p. 30-45.
- Charbonneau, S.M. and I.C. Munro. 1983. Dietary factors affecting pesticide toxicity. In Miyamoto, J. and P.C. Kearney (eds.). *Pesticide chemistry: Human welfare and the environment. Vol.3: Matsunaka, S., D.H. Hudson, and S.D. Murphy (eds.). Mode of action, metabolism, and toxicology.* Pergamon Press, Oxford. 1983.
- Clark, G. 1971. Organophosphate insecticides and behaviour, a review. *Aerospace Med.* 42: 735-740.
- Costa, L.G., B.W. Schwab, and S.D. Murphy. 1982. Tolerance to anticholinesterase compounds in mammals. *Toxicol.* 25: 79-97.
- Crawford, Jr., H.S. and D.T. Jennings. 1982. Relationships of birds and spruce budworms—literature review and annotated bibliography. *For. Serv., U.S. Dept. Agric., Bibliographies and Literature of Agric.* No. 23. 31 p.
- Crick, H.Q.P. and C.J. Spray. 1987. The impact of aerial applications of fenitrothion on forest bird populations. In S.R. Leather, J.T. Stoakley, and H.F. Evans (eds.). *Population biology and control of the pine beauty moth.* Her Majesty's Stationery Office, London, U.K. p. 76-86.
- Dahlgren, J. 1988. Variation in the diet composition: A hazard to the reliability of the LD50-test. *Environ. Pollut.* 49: 177-181.
- Davis, D.E. 1970. Evaluation of techniques for measuring mortality. *J. Wildl. Dis.* 6: 365-375.
- DeWeese, L.R., C.J. Henny, R.L. Floyd, K.A. Bobal, and A.W. Schultz. 1979. Response of breeding birds to aerial sprays of trichlorfon (Dylox) and carbaryl (Sevin-4-oil) in Montana forests. *U.S. Fish Wildl. Serv., Spec. Sci. Rep. Wildl.* 224. 29 p.
- DeWeese, L.R., L.C. McEwen, L.A. Settimi, and R.D. Deblinger. 1983. Effects on birds of fenitrothion aerial application for mosquito control. *J. Econ. Entomol.* 76: 906-911.
- Eidt, D.C. and K.M.S. Sundaram. 1975. The insecticide fenitrothion in headwater streams from large-scale forest spraying. *Can. Entomol.* 107: 735-742.
- Eidt, D.C., A.J. Sosiak, and V.N. Mallet. 1984. Partitioning and short-term persistence of fenitrothion in New Brunswick (Canada) headwater streams. *Arch. Environ. Contam. Toxicol.* 13: 43-52.
- Eidt, D.C. and V.N. Mallet. 1986. Accumulation and persistence of fenitrothion in needles of balsam fir and possible effects on abundance of *Neodiprion abietis* (Harris) (Hymenoptera: Diprionidae). *Can. Entomol.* 118: 69-77.
- Eidt, D.C. and P.A. Pearce. 1986. The biological consequences of lingering fenitrothion residues in conifer foliage - a synthesis. *Forest Chro.* August 1986. p. 246-249.
- Fairbrother, A. and J.K. Bennett. 1988. The usefulness of cholinesterase measurements. *J. Wildl. Dis.* 24 : 587-590.
- Farage-Elawar, M. and B.M. Francis. 1988. The effect of three organophosphorus esters on brain and blood neurotoxic esterase and acetylcholinesterase. *Pest. Biochem. Physiol.* 31: 175- 181.
- Finley, R.B. 1965. Adverse effects on birds of phosphamidon applied to a Montana forest. *J. Wildl. Manage.* 29: 500-591.

- Findlay, G.M., G.J. Howe, and W.L. Lockhart. 1974. Impact of fenitrothion upon Japanese Quail (*Coturnix coturnix japonica*) in a forest ecosystem. *Manit. Entomol.* 8: 10-15.
- Fleming, W.J. 1981. Recovery of brain and plasma cholinesterase activities in ducklings exposed to organophosphorus pesticides. *Arch. Environ. Contam. Toxicol.* 10: 215-229.
- Fleming, W.J. and S.P. Bradbury. 1981. Recovery of cholinesterase activity in mallard ducklings administered organophosphorus pesticides. *J. Toxicol. Environ. Health* 8: 885-897.
- Fleming, W.J. and C.E. Grue. 1981. Recovery of cholinesterase activity in five avian species exposed to dicrotophos, an organophosphorus pesticide. *Pest. Biochem. Physiol.* 16: 129-135.
- Fleming, W.J., G.H. Heinz, J.C. Franson, and B.A. Rattner. 1985. Toxicity of Abate 4E (temephos) in mallard ducklings and the influence of cold. *Environ. Toxicol. Chem.* 4: 193-199.
- Fowle, C.D. 1965. A preliminary report on the effects of phosphamidon on bird populations in central New Brunswick. *Can. Wildl. Serv. Occas. Papers No. 7.* 54 p.
- Fowle, C.D. 1972. Effects of phosphamidon on forest birds in New Brunswick. *Can. Wildl. Serv. Rep. Ser. No. 16.* 25 p.
- Frawley, J.P., R. Weir, T. Tusing, K.B. DuBois, and J.C. Calandra. 1963. Toxicological investigations on Delnav. *Toxicol. Appl. Pharmacol.* 5: 605-624.
- Fudge, S. and Associates Ltd. 1988. The 1987 fenitrothion monitoring study on songbird brain cholinesterase activity and balsam fir foliage. Newfoundland and Labrador Dept. For. Res. Lands, F-128. 65 p.
- Fudge, Lane and Associates Ltd. 1987. A study to monitor the impacts of the 1986 hemlock looper insecticide spray program on songbirds. In *Environmental Monitoring: Hemlock looper spraying program 1986*. Newfoundland and Labrador Dept. Environ. 51 p.
- Gage, S.H. 1968. Consumption by adult birds of the spruce budworm at different larval densities. *Can. For. Res. Lab., Marit. Reg., Intern Rep. M-36.* 5 p.
- Galindo, J.C., R.J. Kendall, C.J. Driver, and T.E. Lacher, Jr. 1985. The effect of methylparathion on susceptibility of Bobwhite Quail (*Colinus virginianus*) to domestic cat predation. *Behav. Neural Biol.* 43: 21-36.
- Giles, R.H. 1970. The ecology of a small forested watershed treated with the insecticide malathion-S35. *Wildl. Monogr.* 24.
- Grue, C.E. 1982. Response of common grackles to dietary concentrations of four organophosphate pesticides. *Arch. Environ. Contam. Toxicol.* 11: 617-626.
- Grue, C.E. and C.C. Hunter. 1984. Brain cholinesterase activity in fledgling starlings: implications for monitoring exposure of songbirds to ChE inhibitors. *Bull. Environ. Contam. Toxicol.* 32: 282-289.
- Grue, C.E. and B.K. Shipley. 1981. Interpreting population estimates of birds following pesticide applications - behaviour of male starlings exposed to an organophosphate pesticide. *Studies in Avian Biology No. 6:* 292-296.
- Grue, C.E. and B.K. Shipley. 1984. Sensitivity of nestling and adult starlings to dicrotophos, an organophosphate pesticide. *Environ. Res.* 35: 454-465.
- Grue, C.E., W.J. Fleming, D.G. Busby, and E.F. Hill. 1983. Assessing hazards of organophosphate pesticides to wildlife. *Trans. 48th N. Amer. Wildl. Nat. Resources Conf.* p. 200-220.
- Grue, C.E., G.V.N. Powell, and N.L. Gladson. 1981. Brain cholinesterase (ChE) activity in nestling starlings: implications for monitoring exposure of nestling songbirds to ChE inhibitors. *Bull. Environ. Contam. Toxicol.* 26: 544-547.
- Grue, C.E., G.V.N. Powell, and C.H. Gorsuch. 1982a. Assessing effects of organophosphates on songbirds: comparison of a captive and free-living population. *J. Wildl. Manage.* 46: 766-768.

- Grue, C.E., G.V.N. Powell, and M.J. McChesney. 1982b. Care of nestlings by wild female starlings exposed to an organophosphate pesticide. *J. Appl. Ecol.* 19: 327-335.
- Haliburton, W., W.W. Hopewell, and W.N. Yule. 1975. Deposit assessment, chemical insecticides. In M.L. Prebble (ed.). *Aerial control of forest insects in Canada*. Dept. Environ., Ottawa. p. 59-67.
- Hamilton, G.A., K. Hunter, and A.D. Ruthven. 1981. Inhibition of brain acetylcholinesterase activity in songbirds exposed to fenitrothion during aerial spraying of forests. *Bull. Environ. Contam. Toxicol.* 27: 856-863.
- Heinz, G.H., E.F. Hill, W.H. Stickel, and L.F. Stickel. 1979. Environmental contaminant studies by the Patuxent Wildlife Research Center. In E.E. Kenaga (ed.). *Avian and mammalian wildlife toxicology*. Am. Soc. Testing Materials Spec. Tech. Pub. 693.
- Hill, E.F. 1971. Toxicity of selected mosquito larvicides to some common avian species. *J. Wildl. Manage.* 35: 757-762.
- Hill, E.F. 1988a. The usefulness of cholinesterase measurements: a response. *J. Wildl. Dis.* 24: 591-593.
- Hill, E.F. 1988b. Brain cholinesterase activity of apparently normal wild birds. *J. Wildl. Dis.* 24: 51-61.
- Hill, E.F. and M.B. Camardese. 1986. Lethal dietary toxicities of environmental contaminants and pesticides to *Coturnix*. U.S. Dept. Int., U.S. Fish Wildl. Serv., Fish Wildl. Tech. Rep. 2. Washington, D.C.
- Hill, E.F. and W.J. Fleming. 1982. Anticholinesterase poisoning of birds: field monitoring and diagnosis of acute poisoning. *Environ. Toxicol. Chem.* 1: 27-38.
- Hill, E.F., R.G. Heath, J.W. Spann, and J.D. Williams. 1975. Lethal dietary toxicities of environmental pollutants to birds. U.S. Fish Wildl. Serv., Spec. Sci. Rep. Wildl. No. 191. 61 p.
- Himel, C.M. and A.D. Moore. 1967. Spruce budworm mortality as a function of aerial spray drop size. *Science* 156: 1250-1251.
- Himel, C.M. and A.D. Moore. 1969. Spray droplet size in the control of spruce budworm, boll weevil, bollworm and cabbage looper. *J. Econ. Entomol.* 62: 916-918.
- Hoffman, D.J. and P.H. Albers. 1984. Evaluation of potential embryotoxicity and teratogenicity of 42 herbicides, insecticides, and petroleum contaminants to mallard eggs. *Arch. Environ. Contam. Toxicol.* 13: 15-27.
- Holmes, S.B. 1988. Behavioural and reproductive correlates of reduced levels of cholinesterase activity in Zebra Finches following acute exposure to fenitrothion. M.Sc.thesis, Queen's University, Kingston, Ontario. 89 p.
- Holmes, S.B., P.D. Kingsbury, G. Mamarbachi, and P. Mathieu. 1984. Distribution of fenitrothion residues in brook trout (*Salvelinus fontinalis*) and lake trout (*Salvelinus namaycush*) tissues following aerial applications to Lac Ste-Anne, Québec. *Bull. Environ. Contam. Toxicol.* 33: 468-475.
- Holmes, S.B., R.L. Millikin, and P.D. Kingsbury. 1981. Environmental effects of a split application of Sevin-2-oil. *Can. For. Serv., For. Pest Manage. Inst. Rep. FPM-X-46*.
- Hudson, R.H., M.A. Haegle, and R.K. Tucker. 1979. Acute oral and percutaneous toxicity of pesticides to mallards: correlations with mammalian toxicity data. *Toxicol. Appl. Pharmacol.* 47: 451-460.
- Hudson, R.H., R.K. Tucker, and M.A. Haegle. 1972. Effects of age on sensitivity: acute oral toxicity of 14 pesticides to mallard ducks of several ages. *Toxicol. Appl. Pharmacol.* 22: 556-561.
- Hudson, R.H., R.K. Tucker, and M.A. Haegle. 1984. Handbook of toxicity of pesticides to wildlife. Second edition. U.S. Fish Wildl. Serv. Resource Publ. 153. Washington, D.C.
- Hunter, M.L. Jr. and J.W. Witham. 1985. Effects of a carbaryl induced depression of arthropod abundance on the behaviour of Parulinae warblers. *Can. J. Zool.* 63: 2612-2616.

- Hurtig, H.H., J.J. Fettes, A.P. Randall, and W.W. Hopewell. 1953. A field investigation of the relation between the amount of DDT spray deposited, the physical properties of the spray, and its toxicity to larvae of the spruce budworm. Defense Research Board of Canada, Suffield Rep. No. 176, Part 6. 141 p.
- Jackson, J.J. and P.O. Park. 1973. The effects of fenthion on a nesting population of *Quelea* during experimental control by aerial spraying. In H.H. Cones Jr. and W.B. Jackson (eds.). Proc. sixth bird control seminar. Bowling Green State University, Bowling Green, Ohio. p. 53-73.
- Joyce, R.J.V. and J. Beaumont. 1978. Collection of spray droplets and chemical larvae, foliage and ground deposition. In A.V. Holden and D. Bevan (eds.). Control of pine beauty moth by fenitrothion in Scotland, 1978. For. Comm. Occas. Paper No. 4. p. 63-80.
- Joyce, R.J.V., G.W. Schaeffer, and K. Allsop. 1981. Distribution of spray and assessment of larval mortality at Annabaglish. In A.V. Holden and D. Bevan (eds.). Aerial application of insecticide against pine beauty moth. For. Comm. Occas. Paper No. 11. p. 15-46.
- Joyce, R.J.V., S. Uk, and C.S. Parkin. 1977. Efficiency in pesticide application. In D.L. Watson and A.W. Brown (eds.). Insecticide management and insecticide resistance. XV International Congress in Entomology, Washington 1976. p. 135-155.
- Julin, A.M. and F.J. Gramlich. 1978. Effects of Orthene® on songbird brain cholinesterase in Maine, 1977. Forest Insect and Disease Management, evaluation report. U.S. Dept. Agric., Broomall, Pa. 19008. 5 p.
- Karczmar, A.G. 1984. Acute and long lasting central actions of organophosphorus agents. Fundamental Appl. Toxicol. 4, S1-S17.
- Keith, J.O. and M.S. Mulla. 1966. Relative toxicity of five organophosphorus mosquito larvicides to mallard ducks. J. Wildl. Manage. 30: 553-563.
- King, K.A., D.H. White, and C.A. Mitchell. 1984. Nest defense behaviour and reproductive success of laughing gulls sublethally dosed with parathion. Bull. Environ. Contam. Toxicol. 33: 499-504.
- Kingsbury, P.D. 1976. A history of the effects of aerial forest spraying in Canada on aquatic fauna. Can. For. Serv., Chem. Control Res. Inst. Rep. CC-X-117.
- Kingsbury, P.D. 1978. Aquatic impact studies by Forest Pest Management Institute in Québec spruce budworm spray block 305, 1977. Can. For. Serv., For. Pest. Manage. Inst. Rep. FPM-X-12.
- Kingsbury, P.D. and B.B. McLeod. 1981. Fenitrothion and forest avifauna studies on the effects of high dosage applications. Can. For. Serv., For. Pest Manage. Inst. Inf. Rep. FPM-X-43.
- Kobayashi, H., A. Yuyama, M. Kudo, and N. Matsusaka. 1983. Effects of organophosphorus compounds, O,O-dimethyl O-(2,2-dichlorovinyl) phosphate (DDVP) and O,O-dimethyl O-(3-methyl 4-nitrophenyl) phosphorothioate (fenitrothion), on brain acetylcholine content and acetylcholinesterase activity in Japanese Quail. Toxicol. 28: 219-227.
- Kulczycki, A. 1975. [Effect of organophosphate insecticides- nogos and owadofos on the survival rate and reproduction in pheasants (*Phasianus colchicus* L.)]. Acta Agrar. Silvestria Ser. Zootech. 15(2): 63-72. [In Polish].
- Kutty, K.M. 1980. Review: biological function of cholinesterase. Clin. Biochem. 13: 239-243.
- Lack, D. 1954. The natural regulation of animal numbers. Oxford University Press, London. 343 p.
- Lauwerys, R.R. and S.D. Murphy. 1969a. Comparison of assay methods for studying O,O-diethyl O-p-nitrophenyl phosphate (paraoxon) detoxication *in vitro*. Biochem. Pharmacol. 18: 789-800.
- Lauwerys, R.R. and S.D. Murphy. 1969b. Interaction

- between paraoxon and tri-o-tolyl phosphate in rats. *Toxicol. Appl. Pharmacol.* 14: 348-357.
- Lehoux, D., R. Ouellet, and P. Laporte. 1982. A comparative study of various methods for monitoring forest bird populations following experimental insecticide sprays. Unpublished manuscript. 92 p.
- Lockhart, W.L., D.A. Metner, and N. Grift. 1973. Biochemical and residue studies on rainbow trout, *Salmo gairdneri*, following field and laboratory exposures to fenitrothion. *Manit. Entomol.* 7: 26-36.
- Ludke, J.L., E.F. Hill, and M.P. Dieter. 1975. Cholinesterase (ChE) response and related mortality among birds fed ChE inhibitors. *Arch. Environ. Contam. Toxicol.* 3: 1-21.
- Maguire, C.C. and B.A. Williams. 1987a. Response of thermal stressed Bobwhite to organophosphorus exposure. *Environ. Pollut.* 47: 25-39.
- Maguire, C.C. and B.A. Williams. 1987b. Cold stress and acute organophosphorus exposure: interaction effects on juvenile Northern Bobwhite. *Arch. Environ. Contam. Toxicol.* 16: 477-481.
- Maguire, R.J. and E.J. Hale. 1980. Distribution of fenitrothion sprayed on a pond: Kinetics of its distribution and transformation in water and sediment. *J. Agric. Food Chem.* 28: 372-378.
- Mallet, V.N. and G. Volpé. 1982. A chemical residue survey in relation to the 1980 spruce budworm spray program in New Brunswick (Canada). *J. Environ. Sci. Health B17*: 71-736.
- Martin, A.D., G. Norman, P.I. Stanley, and G.E. Westlake. 1981. Use of reactivation techniques for the differential diagnosis of organophosphorus and carbamate pesticide poisoning in birds. *Bull. Environ. Contam. Toxicol.* 26: 775-780.
- Mason, B.J. 1971. *The physics of clouds*. 2nd Ed. Clarendon Press, Oxford.
- Matthews, G.A. 1979. *Pesticide application methods*. Longman, New York. 480 p.
- McCracken, A. 1980. Increase efficiency with 'ULV' contact action requires insect contact. *Agr. Aviation.* 21: 11-17.
- McEwen, L.C. 1981. Review of grasshopper pesticides vs. rangeland wildlife and habitat. *Proc. Wildl. Livestock Relationships Symp., For. Wildl. Range Exp. Stn., University of Idaho, Moscow, Idaho.* p. 362-382.
- McEwen, L.C., C.E. Knittle, and M.L. Richmond. 1972. Wildlife effects from grasshopper insecticides sprayed on short-grass range. *J. Range Manage.* 25: 188-194.
- McLeod, J.M. 1967. The effect of phosphamidon on bird populations in jack pine stands in Québec. *Can. Field-Nat.* 81: 102-106.
- McNeil, J.N., R. Greenhalgh, and J.M. McLeod. 1979. Persistence and accumulation of fenitrothion residues in jack pine foliage and their effect on the Swaine jack pine sawfly, *Neodiprion swainei* (Hymenoptera: Diprionidae). *Environ. Entomol.* 8: 752-755.
- Mehrotra, K.N., Y.P. Beri, S.S. Misra, and A. Phokela. 1967. Physiological effects of malathion on the house sparrow. *Indian J. Exp. Biol.* 5: 219-221.
- Mihara, K., Y. Misaki, and J. Miyamoto. 1979. Metabolism of fenitrothion in birds. *J. Pestic. Sci.* 4: 175-185.
- Millikin, R.L. 1987. Sublethal effects of fenitrothion on forest passerines. M.Sc. thesis, University of British Columbia, Vancouver, British Columbia. 126 p.
- Mineau, P. and D.B. Peakall. 1987. An evaluation of avian impact assessment techniques following broad-scale forest insecticide sprays. *Environ. Toxicol. Chem.* 6: 781-791.
- Mineau, P. and B.T. Collins. 1988. Avian mortality in agro-ecosystems. 2. Methods of detection. In M.P. Greaves, B.D. Smith, and P.W. Greig-Smith (eds.). *Symposium on field methods for the study of environmental effects of pesticides*. British Crop Protection Council. Croydon, U.K. p. 13-27.
- Mitchell, C.A. and D.H. White. 1982. Seasonal brain

- acetylcholinesterase activity in three species of shorebirds overwintering in Texas. *Bull. Environ. Contam. Toxicol.* 29: 360-365.
- Miyamoto, J. 1977. Degradation of fenitrothion in terrestrial and aquatic environments including photolytic and microbial reactions. In J.R. Roberts, R. Greenhalgh, and W.K. Marshall (eds.). *Proc. symp. on fenitrothion: The long-term effects of its use in forest ecosystems*. NRCC Publ. No. NRCC 16073, Ottawa. p. 105-134.
- Monenco Ltd.. 1978. Assessment of the effects of the 1977 New Brunswick spruce budworm control program on fish food organisms and fish growth. Rep. to New Brunswick Dept. Nat. Resources. 40 p., appendices.
- Moody, R.P. 1978. The persistence, transport and metabolism of fenitrothion in conifers. M.Sc. thesis, University of Ottawa, Ottawa. 76 p.
- Moody, R.P., R. Greenhalgh, L. Lockhart, and P. Weinberger. 1978. The fate of fenitrothion in an aquatic ecosystem. *Bull. Environ. Contam. Toxicol.* 19: 8-14.
- Moody, R.P., R. Prasad, R. Greenhalgh, and P. Weinberger. 1976. Fate of fenitrothion in forest trees. VII. Persistence, translocation and metabolism of ¹⁴C-fenitrothion in conifers. *Can. For. Serv., Chem. Control Res. Inst. Rep.* CC-X-112.
- Moody, R.P., R. Prasad, R. Greenhalgh, and P. Weinberger. 1977. Translocation of ring labeled ¹⁴C-fenitrothion in conifers. *Proc. symp. on fenitrothion: The long-term effects of its use in forest ecosystems*. J.R. Roberts, R. Greenhalgh, and W.K. Marshall (eds.). NRCC Publ. No. NRCC 16073, Ottawa. p. 218-231.
- Mook, L.J. 1963. Birds and spruce budworm. In R.F. Morris (ed.). *The dynamics of epidemic spruce budworm populations*. *Mem. Entomol. Soc. Can.* 31: 268-271.
- Morgan, J.H. 1968. Preliminary report on the uptake of P32 Sumithion by adult house sparrows. *Can. Wildl. Serv., Pestic. Section Manuscript Rep. No.* 11. 4 p.
- Morin, R., G. Gaboury, G. Mamarbachi, and R. Dostie. 1983. Résidus de fénitrothion et d'aminocarbe dans l'environnement. Ministère de l'Énergie et des Ressources du Québec. 60 p.
- Morin, R., G. Gaboury, and R. Dostie. 1984. Résidus de fénitrothion et d'aminocarbe dans l'environnement. Ministère de l'Énergie et des Ressources du Québec. 56 p.
- Morris, R.F. 1963. Predation and the spruce budworm. In R.F. Morris (ed.). *The dynamics of epidemic spruce budworm populations*. *Mem. Entomol. Soc. Can.* 31: 244-248.
- Morris, R.E., W.F. Cheshire, C.A. Miller, and D.G. Mott. 1958. The numerical response of avian and mammalian predators during a gradation of the spruce budworm. *Ecology* 39: 487-494.
- Morrison, B.R.S. and D.E. Wells. 1981. The fate of fenitrothion in a stream environment and its effect on the fauna following aerial spraying of a Scottish forest. *Sci. Total Environ.* 19: 233-252.
- Moulding, J.D. 1976. Effects of a low-persistence insecticide on forest bird populations. *Auk* 93: 692-708.
- Murphy, S.D. 1975. Pesticides. In L.J. Casarett and J. Doull (eds.). *Toxicology: The basic science of poisons*. MacMillan, New York. p. 408-453.
- National Research Council Canada. 1975. Fenitrothion: The effects of its use on environmental quality and its chemistry. NRCC Publ. No. NRCC 14104. Ottawa. 162 p.
- National Research Council Canada. 1977. Fenitrothion: The long-term effects of its use in forest ecosystems - Current Status. NRCC Publ. No. NRCC 15389. Ottawa. 18 p.
- Nguyen, N.T. and P.M. Symons. 1984. Aerial spraying of wheat; a comparison of conventional low volume with ultra low volume spraying. *Pestic. Sci.* 15: 337-343.
- Nigam, P.C. 1975. Chemical insecticides. In M.L.

- Prebble (ed.). Aerial control of forest insects in Canada. Dept. Environ., Ottawa. p. 8-24.
- O'Brien, R.D. 1960. Toxic phosphorus esters: Chemistry, metabolism and biological effects. Academic Press, New York. 434 p.
- O'Brien, R.D. 1967. Insecticides: action and metabolism. Academic Press, New York. 332 p.
- Peakall, D.B. and J.R. Bart. 1983. Impacts of aerial application of insecticides on forest birds. CRC Crit. Rev. Environ. Control. 13: 117-165.
- Pearce, P.A. 1967. Summary statement on the effects on bird populations of Novathion, Sumithion, and Sumithion-Phosphamidon mixtures, New Brunswick, 1967. Can. Wildl. Serv., Pestic. Section Manuscript Rep. No. 7. 39 p.
- Pearce, P.A. 1968. Effects on bird populations of Phosphamidon and Sumithion used for spruce budworm control in New Brunswick and hemlock looper control in Newfoundland in 1968: a summary statement. Can. Wildl. Serv., Pestic. Section Manuscript Rep. No. 14. 56 p.
- Pearce, P.A. 1971. Side effects of forest spraying in New Brunswick. Trans. 36th N. Amer. Wildl. Nat. Resources Conf. p. 163-170.
- Pearce, P.A., G.L. Brun, and J. Witteman. 1979b. Off-target fall-out of fenitrothion during 1978 forest spraying operations in New Brunswick. Bull. Environ. Contam. Toxicol. 23: 503-508.
- Pearce, P.A. and D.G. Busby. 1980. Research on the effects of fenitrothion on the White-throated Sparrow. In I.W. Varty (ed.). Environmental surveillance in New Brunswick, 1978-79: effects of spray operations for forest protection against spruce budworm. Committee for environmental monitoring of forest insect control operations. University of New Brunswick. p. 24-27.
- Pearce, P.A. and N.R. Garrity. 1981. Impact of aminocarb (MatacilR) spraying on forest songbirds in northern New Brunswick. Can. Wildl. Serv. Progr. Notes 121: 17 p.
- Pearce, P.A. and D.B. Peakall. 1977. The impact of fenitrothion on bird populations in New Brunswick. In J.R. Roberts, R. Greenhalgh, and W.K. Marshall (eds.). Proc. symp. on fenitrothion: The long-term effects of its use in forest ecosystems. NRCC Publ. No. NRCC 16073. Ottawa. p. 299-305.
- Pearce, P.A., D.B. Peakall, and A.J. Erskine. 1976. Impact on forest birds of the 1975 spruce budworm spray operation in New Brunswick. Can. Wildl. Serv. Progr. Notes 62: 7 p.
- Pearce, P.A., D.B. Peakall, and A.J. Erskine. 1979a. Impact on forest birds of the 1976 spruce budworm spray operation in New Brunswick. Can. Wildl. Serv. Progr. Notes 97: 15 p.
- Peters, D.C. 1979. Growth rates of White-throated Sparrow chicks on a fenitrothion sprayed and an unsprayed plot in northeastern New Brunswick. B.Sc.F. thesis, University of New Brunswick, Fredericton, New Brunswick. 22 p.
- Peterson, J.W. 1969. The effects of Sumithion on birds. In A cooperative pilot study of the insecticide Sumithion. U.S.D.A. Forest Service; U.S.D.I. Bureau of Sport Fish. Wildl.; Maine State Dept. Forestry. p. 12-19.
- Peterson, J.W. 1971. The effects of Accothion on birds and other fauna. In Environmental studies of Accothion for spruce budworm control. A cooperative study by the State of Maine, U.S.D.A. and U.S.D.I. p. 81-87.
- Peterson, J.W. 1976. Effects of Dylox, Matacil and Sumithion on birds, small mammals and amphibians. In 1975 cooperative pilot control project of Dylox, Matacil, and Sumithion for spruce budworm control in Maine. Forest Insect and Disease Management; Northeastern Area State and Private Forestry; U.S. Forest Service. p.2-10.
- Pope, G.G. and P. Ward. 1972. The effects of small applications of an organophosphorus poison, fenthion, on the weaverbird, *Quelea quelea*. J. Pestic. Sci. 3: 197-205.
- Powell, G.V.N. 1984. Reproduction by an altricial songbird, the Red-winged Blackbird, in fields treated with the organophosphate insecticide fenthion. J. Appl. Ecol. 21: 83-95.
- Powell, G.V.N. and D.C. Gray. 1980. Dosing free-living nestling starlings with an

- organophosphate pesticide, famphur. J. Wildl. Manage. 44: 918-921.
- Prasad, R. and R.P. Moody. 1976. Fate of fenitrothion in forest trees. VIII. Persistence, translocation and metabolism of "C-fenitrothion in Jack Pine (*Pinus banksiana*, Lamb.) in relation to sawfly mortality. Can. For. Serv., Chem. Control Res. Inst. Inf. Rep. CC-X-126.
- Prebble, M.L. 1975. Introduction. In M.L. Prebble (ed.). Aerial control of forest insects in Canada. Dept. Environ., Ottawa. p. 1.
- Quantick, H.R. 1985. Aviation in crop protection, pollution, and insect control. Collins, London. 428 p.
- Ralph, C.J. and J.M. Scott (eds.). 1981. Estimating numbers of terrestrial birds. Stud. Avian Biol. 6. 630 p.
- Rattner, B.A. 1982. Diagnosis of anticholinesterase poisoning in birds: effects of environmental temperature and underfeeding on cholinesterase activity. Environ. Toxicol. Chem. 1: 329-335.
- Rattner, B.A., J.M. Becker, and T. Nakatsugawa. 1987. Enhancement of parathion toxicity to quail by heat and cold exposure. Pestic. Biochem. Physiol. 27: 330-339.
- Rattner, B. A., L. Sileo, and C.G. Scanes. 1982. Hormonal responses and tolerance to cold of female quail following parathion ingestion. Pestic. Biochem. Physiol. 18: 132-138.
- Richmond, M. L., C.H. Henny, R.L. Floyd, R.W. Mannan, D.M. Finch, and L.R. DeWeese. 1979. Effects of Sevin-4-oil, Dimilin, and Orthene on forest birds in northeastern Oregon. Pacific Southwest Forest and Range Exp. Stn., Forest Service, U.S. Dept. Agric., Berkeley, Calif. Res. Paper PSW-148. 19 p.
- Ricklefs, R.E. 1979. Ecology. 2nd edition. Chiron Press, New York. 966 p.
- Robertson, J.L. and L.M. Boetler. 1979. Toxicity of insecticides to Douglas fir tussock moth (*Orgyia pseudotsugata*) Lepidoptera: Lymantriidae. Can. Entomol. 111: 1161-1175.
- Sanders, C.J. 1970. Populations of breeding birds in the spruce-fir forest of northwestern Ontario. Can. Field-Nat. 84: 131-135.
- Schafer, E.W. 1972. The acute oral toxicity of 369 pesticidal, pharmaceutical and other chemicals to wild birds. Toxicol. Appl. Pharmacol. 21: 315-330.
- Schafer, E.W. and R.B. Brunton. 1979. Indicator bird species for toxicity determinations: Is the technique usable in test method development? Am. Soc. Testing Materials, Spec. Tech. Pub. 680.
- Sexsmith, W. 1987. The evolution of buffer zones for forest insect spraying in New Brunswick. In Buffer zones: Their application to forest insect control operations. J.G. Trial (ed.). Proc. Buffer Zone Workshop, Eastern Spruce Budworm Council's Environmental Committee, Québec City, P.Q., April 1986. p. 4-12.
- Shirazi, M.A., R.S. Bennett, and LeVaughn Lowrie. 1988. An approach to environmental risk assessment using avian toxicity tests. Arch. Environ. Contam. Toxicol. 16: 263-271.
- Spillman, J.J. and R.J.V. Joyce. 1978. Low-volume and ultra-low-volume spray trials from aircraft over Thetford Forest. In A.V. Holden and D. Bevan (eds.). Control of pine beauty moth by fenitrothion in Scotland 1978. For. Comm. p. 31-39.
- Spray, C.J., H.Q.P. Crick, and A.D.M. Hart. 1987. Effects of aerial applications of fenitrothion on bird populations of a Scottish pine plantation. J. Appl. Ecol. 24: 29-47.
- Stickel, L. 1964. Wildlife studies, Patuxent Wildlife Research Center. In Pesticide-wildlife studies 1963: A review of Fish and Wildlife Service investigations during the calendar year. U.S. Dept. Interior, Fish Wildl. Serv. Circ. 199: 77-115.
- Stickel, W.H. 1974. Effects on wildlife of newer pesticides and other pollutants. Proc. Ann. Conf. West. Assoc. Game Fish Comm. 53: 484-491.
- Stromborg, K.L. 1981. Reproductive tests of diazinon on bobwhite quail. In Avian and mammalian wildlife toxicology: second conference. Am. Soc. Testing Materials, 757. p. 19-31.

- Stromborg, K.L., W.N. Beyer, and E. Kolbe. 1982. Diazinon residues in insects from sprayed tobacco. *Chem. in Ecol.* 1: 93-97.
- Stromborg, K.L., C.E. Grue, J.D. Nichols, G.R. Hepp, J.E. Hines, and H.C. Bourne. 1988. Postfledging survival of European Starlings exposed as nestlings to an organophosphorus insecticide. *Ecology* 69: 590-601.
- Strong, K.W. and J. Wells. 1987. A study to monitor the impact of the 1985 hemlock looper insecticide spray program on songbird populations. In *Environmental Monitoring: Hemlock looper spraying program 1985*. Newfoundland and Labrador Dept. Environ. 26 p.
- Stutzenbaker, C.D., K. Brown, and D. Lobpries. 1984. Special report: an assessment of the accuracy of documenting water fowl die-offs in a Texas coastal marsh. *Proc. symposium on lead poisoning in wild waterfowl: a workshop. A contribution of federal aid in wildl. restoration project W-106-R, Texas Parks and Wildl. Dept.* p. 88-95.
- Sundaram, K.M.S. 1973. Degradation dynamics of fenitrothion insecticide in aquatic environments associated with forest spraying. *Proc. symposium on water quality parameters*. Burlington, Ontario. *Chem. Inst. Can.* 21 p.
- Sundaram, K.M.S. 1974a. Distribution and persistence of fenitrothion residues in foliage, soil and water in Larose Forest. *Can. For. Serv., Chem. Control Res. Inst. Inf. Rep. CC-X-64*.
- Sundaram, K.M.S. 1974b. Persistence studies of insecticides: III. Accumulation of fenitrothion and its oxygen analog in foliage, soil and water in Larose Forest. *Can. For. Serv., Chem. Control Res. Inst. Inf. Rep. CC-X-65*.
- Sundaram, K.M.S. 1984a. Fenitrothion residues in some forestry samples from a plantation forest. *Can. For. Serv., For. Pest Manage. Inst. Inf. Rep. FPM-X-66*.
- Sundaram, K.M.S. 1984b. Residue levels of fenitrothion and aminocarb in the air samples collected from experimental spray blocks in New Brunswick in 1982. *J. Environ. Health B19*: 409-426.
- Sundaram, K.M.S. 1986. A comparative evaluation of dislodgeable and penetrated residues, and persistence characteristics of aminocarb and fenitrothion, following application of several formulations onto conifer trees. *J. Environ. Health B21*: 539-560.
- Sundaram, K.M.S. 1987. Persistence characteristics of operationally sprayed fenitrothion in nearby unsprayed areas of a conifer forest ecosystem in New Brunswick. *J. Environ. Sci. Health B22*: 413-438.
- Sundaram, K.M.S. and R. Nott. 1984. Fenitrothion residues in selected components of a conifer forest following aerial application of tank mixes containing Triton X-100. *Can. For. Serv., For. Pest Manage. Inst. Inf. Rep. FPM-X-65*.
- Sundaram, K.M.S. and R. Nott. 1985. Mexacarb residues in selected components of a conifer forest following aerial applications of oil-based and aqueous spray formulations. *J. Environ. Sci. Health B20(4)*: 425-444.
- Sundaram, K.M.S. and R. Nott. 1986. Persistence of fenitrothion residues in a conifer forest environment. *Can. For. Serv., For. Pest Manage. Inst. Inf. Rep. FPM-X-75*.
- Sundaram, K.M.S. and A. Sundaram. 1981. Effect of additives on the persistence of aminocarb in conifer foliage. *Can. For. Serv. Res. Notes* 1: 18-21.
- Sundaram, K.M.S. and A. Sundaram. 1982. Influence of formulation on droplet size, deposit concentration and persistence of fenitrothion in conifers following a simulated aerial application. *Can. For. Serv. Res. Notes* 2: 2-5.
- Sundaram, K.M.S. and A. Sundaram. 1987. Influence of spray deposit patterns, dislodgeable and penetrated residues, and persistence characteristics of fenitrothion in conifer needles. *Pestic. Sci.* 18: 259-271.
- Sundaram, K.M.S. and S.Y. Szeto. 1984. Persistence of aminocarb in balsam fir *Abies balsamea*

- foliage, forest litter, and soil after aerial application of 3 formulations. *J. Agric. Food Chem.* 32: 1138-1141.
- Sundaram, A., K.M.S. Sundaram, and B.L. Cadogan. 1985a. Influence of formulation properties on droplet spectra and soil residues of aminocarb aerial sprays in conifer forests. *J. Environ. Sci. Health B20*: 167-186.
- Sundaram, A., K.M.S. Sundaram, B.L. Cadogan, R. Nott, and J.W. Leung. 1985b. An evaluation of physical properties, droplet spectra, ground deposits and soil residues of aerially applied aminocarb and fenitrothion emulsions in conifer forests in New Brunswick. *J. Environ. Sci. Health B20*: 665-688.
- Sundaram, A. and A. Retnakaran. 1987. Influence of formulation properties on droplet size spectra and ground deposits of aerially-applied pesticides. *Pestic. Sci.* 20: 241-257.
- Svesson, S. 1970. An international standard for a mapping method in bird census work recommended by the International Bird Census Committee. *Audubon Field Notes* 24 (Suppl.) 6: 722-726.
- Symons, P.E.K. 1977. Dispersal and toxicology of the insecticide fenitrothion. *Residue Rev.* 38: 1-36.
- Tucker, R.K., and D.G. Crabtree. 1970. *Handbook of Toxicity of Pesticides to Wildlife*. Denver Wildlife Research Center Publ. 84.
- Tucker, R.K. and M.A. Haegle. 1971. Comparative acute oral toxicity of pesticides to six species of birds. *Toxicol. Appl. Pharmacol.* 20: 57-65.
- Verner, J. 1985. Assessment of counting techniques. In F. Richardson (ed.). *Current Ornithology*. RPlenum Press, New York. p. 247-302.
- von Haartman, L. 1953. Was reizt den Trauerfliegenschapper (*Muscicapa hypoleuca*) zu futtern? *Die Vogelwarte* 16: 157-164.
- Ware, G.W., W.P. Cahill, P.D. Gerhardt, and J.M. Witt. 1970. Pesticide drift, part 4. On target deposits from aerial application of insecticides. *J. Econ. Entomol.* 63: 1982-1983.
- Ware, G.W., B.J. Estes, W.P. Cahill, and K.R. Frost. 1972. Pesticide drift, part 6. Target and drift deposits vs. time of applications. *J. Econ. Entomol.* 65: 1170-1172.
- Welsh, D.A. and D.R. Fillman. 1980. The impact of forest cutting on boreal bird populations. *Am. Birds* 34: 84-94.
- Welty, J.C. 1975. *The life of birds*. 2nd Ed. W.B. Saunders Co., Toronto. 623 p.
- Westlake, E.W., M. Ittig, D.E. Ott, and F.A. Gunther. 1973. Persistence of residues of the insecticide phosphamidon on and in oranges, lemons, and grapefruit, and on and in orange leaves and in dried citrus pulp cattle feed. *J. Agric. Food Chem.* 21: 846-850.
- White, D.H., C.A. Mitchell, and R.M. Prouty. 1983a. Nesting biology of Laughing Gulls in relation to agricultural chemicals in south Texas, 1978-1981. *Wilson Bull.* 95: 540-551.
- White, D.H., C.A. Mitchell, and E.F. Hill. 1983b. Parathion alters incubation behavior of Laughing Gulls. *Arch. Environ. Contam. Toxicol.* 31: 93-97.
- White, D.H., K.A. King, C.A. Mitchell, E.F. Hill, and T.G. Lamont. 1979. Parathion causes secondary poisoning in a Laughing Gull breeding colony. *Bull. Environ. Contam. Toxicol.* 23: 281-284.
- Wood, G.W. and D.K.R. Stewart. 1976. Fenitrothion residues in blueberry fields after aerial forest spraying. *Bull. Environ. Contam. Toxicol.* 15: 623-629.
- Yates, W.E. and N.B. Akesson. 1973. Reducing pesticide chemical drift. In W. Van Valkenburg (ed.). *Pesticide Formulations*. Marcel Dekker, New York. p. 275-341.
- Yule, W.N. 1974. The persistence and fate of fenitrothion insecticide in a forest environment. II. Accumulation of residues in balsam fir foliage. *Bull. Environ. Contam. Toxicol.* 12: 249-252.
- Yule, W.N. and A.F.W. Cole. 1969. Measurement of

- insecticide drift in forestry operations. Proc. 4th Int. Agric. Aviat. Congr., Kingston. p. 346-353.
- Yule, W.N., A.F.W. Cole, and I. Hoffman. 1971. A survey for atmospheric contamination following forest spraying with fenitrothion. Bull. Environ. Contam. Toxicol. 6: 289-296.
- Yule, W.N. and J.R. Duffy. 1972. The persistence and fate of fenitrothion insecticide in a forest environment. Bull. Environ. Contam. Toxicol. 8: 10-17.
- Yule, W.N. and I.W. Varty. 1975. The persistence and fate of fenitrothion insecticide in a forest environment. III. Deposit and residue studies with black spruce and red maple. Bull. Environ. Contam. Toxicol. 13: 678-680.
- Zach, R. 1988. Growth-curve analysis: a critical reevaluation. Auk 105: 208-210.
- Zinkl, J.G., C.J. Henny, and P.J. Shea. 1979. Brain cholinesterase activities of passerine birds sprayed with cholinesterase inhibiting insecticides. In Animals as monitors of environmental pollutants. National Academy of Sciences, Washington, D.C. p. 356-365.
- Zinkl, J.G., C.J. Henny, and L.R. DeWeese. 1977. Brain cholinesterase activities of birds from forests sprayed with trichlorfon (Dylox) and carbaryl (Sevin-4-oil). Bull. Environ. Contam. Toxicol. 17: 379-386.
- Zinkl, J.G., R.B. Roberts, C.J. Henny, and D.J. Lenhart. 1980. Inhibition of brain cholinesterase activity in forest birds and squirrels exposed to aerially applied acephate. Bull. Environ. Contam. Toxicol. 24: 676-683.
- Zinkl, J.G., R.B. Roberts, P.J. Shea, and J. Lasmanis. 1981. Toxicity of acephate and methamidophos to Dark-eyed Juncos. Arch. Environ. Contam. Toxicol. 10: 185-192.

3.13 APPENDICES

Appendix 3.1 Mean and range of brain ChE depression of White-throated Sparrows in relation to fate after dietary exposure to fenitrothion (D. Forsyth, unpub. data)¹.

Concentration of fenitrothion (ppm)	Fate	% Brain ChE depression	
		Mean	Range
200	100% mortality	67.2	56.9 - 75.4
90	100% mortality	68.3	61.7 - 76.3
60	60% mortality	75.3	70.4 - 81.3
60	40% survival	49.4	47.9 - 52.7
40	20% mortality	66.1	66.1
40	80% survival	53.5	47.2 - 58.0
25	100% survival	48.4	41.3 - 54.8
10	100% survival	27.3	3.8 - 52.7

¹ 10 birds per diet except 6 birds for diet containing 10 ppm.

Appendix 3.2 Fenitrothion residues in moss capsules and arthropods collected in area which received two sprays of 210 g a.i./ha fenitrothion (Canadian Wildlife Service, unpublished data).

Organism	Fenitrothion residues (ppm wet weight)	
	Spray 1	Spray 2
Moss capsules	7.37	2.41
	10.8	6.16
	8.78	1.92
	10.8	2.05
	6.85	0.13
	6.16	6.45
	6.00	7.47
		5.93
		0.58
		6.29
Substrate Insects	0.78	0.48
	0.11	0.01
	0.02	0.02
	5.32	0.07
	0.02	0.12
	0.17	0.06
		0.03
Flying Insects		0.02
	0.26	0.14
	0.71	0.18
	0.36	0.24
	0.26	0.41
	17.0	0.17
	0.44	0.32
		1.40
Spiders		0.11
	0.37	0.31
	2.81	0.59
	0.21	1.44
	0.09	0.09
	1.97	0.27
	0.04	0.07
		0.24

CHAPTER FOUR

FENITROTHION EFFECTS ON AQUATIC ORGANISMS

W. L. Fairchild¹, W. R. Ernst², and V. N. Mallet³

¹ Fisheries and Oceans
Freshwater Institute
Winnipeg, Manitoba

² Environment Canada
Conservation and Protection
Dartmouth, Nova Scotia

³ Département de chimie et biochimie
Université de Moncton
Moncton, New Brunswick

TABLE OF CONTENTS

LIST OF TABLES	111
4.0 SUMMARY	113
4.1 INTRODUCTION	114
4.1.1 Overview and Objectives	114
4.1.2 Background and Definitions	115
4.1.3 Considerations for Interpretation of Results	115
4.1.4 Aquatic Buffer Zones	116
4.2 CHEMICAL ANALYSIS	116
4.3 FATE OF FENITROTHION IN AQUATIC SYSTEMS	117
4.3.1 Water	117
4.3.2 Aquatic Plants	118
4.3.3 Sediments	118
4.4 EXPOSURE IN THE AQUATIC ENVIRONMENT	119
4.4.1 Factors Affecting Exposure	119
4.5 AQUATIC RESIDUES DETECTED BY MONITORING PROGRAMS	119
4.6 TOXICITY OF FENITROTHION TO AQUATIC INVERTEBRATES	120
4.6.1 Mode of Action	120
4.6.2 Aquatic Invertebrate Sensitivity	120
4.7 EFFECTS ON INVERTEBRATE DRIFT	120
4.7.1 Characterizing Drift	120
4.7.2 Measuring Drift	125
4.7.3 Effects of Fenitrothion on Drift	125
4.8 EFFECTS ON BENTHIC POPULATIONS	126
4.8.1 Characteristics of Benthic Populations	126
4.8.2 Measurement of Benthic Populations	126
4.8.3 Fenitrothion Effects on Benthos	130
4.9 PROBABILITY OF MEASURABLE DISTURBANCE	138
4.10 LABORATORY STUDIES WITH FISH	139
4.10.1 Acute Toxicity	139
4.10.2 Bioaccumulation	139
4.10.3 Water Quality and Toxicity	144
4.10.4 Sublethal Effects	144
4.10.4.1 Metabolism	144
4.10.4.2 Acetylcholinesterase inhibition	144

4.10.4.3 Biochemical/histopathological changes	144
4.10.4.4 Reproductive effects	146
4.10.4.5 Behavioural effects	148
4.11 FIELD STUDIES WITH FISH	148
4.11.1 Direct Mortality	149
4.11.2 Bioaccumulation	149
4.11.3 Acetylcholinesterase Inhibition	150
4.11.4 Population Effects	150
4.12 INFORMATION DEFICIENCIES	154
4.13 LITERATURE CITED	155

LIST OF TABLES

4.1 Known degradation products of fenitrothion in environmental water.
4.2 Concentration of fenitrothion residues in lotic water after operational aerial spraying.
4.3 Concentration of fenitrothion residues in lentic water after operational aerial spraying.
4.4 Concentration of fenitrothion residues in sediments after operational aerial spraying.
4.5 Summary of concentrations of fenitrothion residues detected in water and sediment after operational aerial spraying.
4.6 Concentration of fenitrothion residues in biota other than fish after operational aerial spraying.
4.7 Effects of fenitrothion on aquatic invertebrate drift after aerial spraying, based on insecticide dosage rate (with no corollary insecticide residue analysis).
4.8 Effects of fenitrothion on aquatic invertebrate drift after aerial spraying (with corollary insecticide residue analysis).
4.9 Effects of fenitrothion on aquatic invertebrate drift in field experiments, based on dosage rate and insecticide residue analysis.
4.10 Effects of fenitrothion on benthic invertebrates in lotic ecosystems after aerial spraying, based on dosage rate (with no corollary insecticide residue analysis).
4.11 Effects of fenitrothion on benthic invertebrates in lotic ecosystems after aerial spraying, based on residue analysis.

- 4.12 Effects of fenitrothion on benthic invertebrate in field experiments based on dosage rate (with no corollary insecticide residue analysis).
- 4.13 Effects of fenitrothion on benthic invertebrates in field experiments, based on insecticide residue analysis.
- 4.14 Acute toxicity of fenitrothion to various warmwater fish species exposed under static conditions.
- 4.15 Acute toxicity of fenitrothion formulations and components of formulations to various coldwater fish species.
- 4.16 Acute toxicity of fenitrothion to various life stages of fish.
- 4.17 Toxicity of various fenitrothion degradation products to fish
- 4.18 Bioconcentration factors (BCF) in fish exposed to fenitrothion under laboratory conditions.
- 4.19 Reproductive effects observed in fish exposed to fenitrothion under static conditions
- 4.20 Effects on development of fish embryos when exposed to fenitrothion.
- 4.21 Concentration of fenitrothion in fish after aerial spray programs.

4.0 SUMMARY

Contamination of natural water bodies by fenitrothion is an unavoidable aspect of forest spraying. Maximum recorded concentrations of fenitrothion in moving (lotic) water after operational spraying range from approximately 1.3-127 $\mu\text{g/L}$ while those in still (lentic) water range from approximately 0.1-1114 $\mu\text{g/L}$.

Estimations of half-lives for fenitrothion in lotic and lentic systems are approximately 13 h and 18 h, respectively. Processes such as hydrolysis, volatilization, and biological degradation, as well as adsorption and absorption to suspended particles and other aquatic substrates, are important pathways for the disappearance of fenitrothion from the water column.

Aquatic biota do absorb and accumulate fenitrothion above ambient concentrations. However, for the most part, residues persist for only short periods of time. Recent studies have indicated that although aqueous fenitrothion residues in bog ponds fall to below detectable levels within several days of spraying, they return to measurable levels the following spring. That indicates that some component of the ecosystem acts as a sink for fenitrothion; and that reservoir is subsequently mobilized.

If fenitrothion reaches a high enough concentration in aquatic systems, it can produce measurable effects on native invertebrates. Increased drift, especially of particularly sensitive groups, is a common consequence of fenitrothion spraying in unprotected streams. The effect of increased drift, even if composed largely of dead insects, is generally not measurable within the benthic invertebrate community; however, a limited number of studies have recorded benthic biomass reductions. None of the studies has been sensitive enough to detect subtle changes in community structure or the elimination of rare or susceptible species. Based on past monitoring studies, the proportion of habitat affected can be estimated, although that does not necessarily reflect what is happening under current use-patterns. Due to improved spraying practices, effects should currently be less than those summarized by reviewers in the late 1970s and judged at that time to be environmentally tolerable. That does not mean that there will be no aquatic effects. Instances of decreased numbers of benthic invertebrates after spraying can still be recorded, but they are likely to be fewer than identified in older literature. However, indications are that invertebrates in small bodies of standing

water (which have been little studied) are sensitive to both past and present operational spray practices. A very recent study indicates that energy flows, and thus ecological processes, in acidic bog ponds are altered by fenitrothion spraying through a shift in the benthic invertebrate community composition.

A sufficiently high concentration of fenitrothion in water, like virtually any other pesticide, is acutely lethal to fish. Native North American fish species have a range of sensitivities to fenitrothion. However, the lowest LC50s are approximately 30 times the immediate maximum concentrations measured after operational forestry spraying. Therefore, the probability of spray-induced mortality, under normal conditions, is remote.

The physical and chemical characteristics of fenitrothion allow fish to concentrate it above ambient levels, resulting in some surprisingly high tissue concentrations immediately after operational forestry spray programs. Measured tissue concentrations in fish have ranged from 0.13-3.7 $\mu\text{g/g}$ wet weight. Tissue residues are generally highest in the viscera, primarily the liver, and lowest in the edible muscle portion. Metabolic kinetics have not been empirically determined. Parent fenitrothion is cleared rapidly however, with measured half-lives being less than 3 d.

The primary mechanism for fish toxicity, as with other organophosphorus insecticides, is through acetylcholinesterase (AChE) inhibition. Brain is the tissue used most frequently for AChE measurement because the variability in brain AChE activity is less than with other tissues. Unfortunately, no clear relationship has been determined between physiological impairment in fish and the degree of brain AChE depression caused by fenitrothion or any other pesticide.

Some biochemical changes, indicative of increased anaerobic metabolism and stress-induced hyperglycemia, have been measured in fish exposed to fenitrothion. However, exposure concentrations in those tests were near lethal levels. At similar concentrations, both tissue and whole animal oxygen consumption were inhibited. Moderately high (0.75 mg/L) fenitrothion concentration during long exposures (60 d) also reduced protein-bound plasma iodine suggesting a thyroid-mediated effect on growth.

Only a few studies report histopathological changes in fish exposed to fenitrothion. One study documented cellular damage to liver, kidney, and intestine but

exposure concentrations (1.0mg/L) and durations (45-90 d) were not environmentally relevant.

During exposures at concentrations exceeding 0.25 mg/L for 30 d and longer, fenitrothion has definite adverse effects on fish reproduction. The effects noted have primarily been a decrease in the gonadosomatic index, an inhibition of gamete development, and a decrease in the number of live young born to viviparous fish.

Fenitrothion is capable of damaging growing fish embryos. Effects documented in the laboratory include spinal deformities, damage to circulatory organs, and reduction in hatch rate. An exposure concentration of 0.25 mg/L for 55-74 h was the threshold which produced spinal defects. In streams draining operational fenitrothion spray blocks in Japan, which were treated with an estimated 1.84 kg a.i./ha, fish embryos had vertebral centrum deformities at rates three to six times those of control fish from streams in untreated areas.

Fenitrothion affects fish behaviour. Measured behavioural changes in salmonids include: a reduction in swimming velocities; a reduction in learning ability; a reduction in ability to avoid predatory attacks; a decrease in tendency to defend territories; a reduction in feeding activities; a reduction of social behaviours; and an increase in coughing. While all or some of those effects could result in a lack of fitness and therefore cause population impacts, the exposure concentrations necessary to elicit such responses were close to lethal levels and at least ten times greater than maximum residues detected in water courses after operational spray programs. Furthermore, it has been determined that goldfish are able to sense and avoid water contaminated with fenitrothion at a very low concentration (0.01 mg/L), and that salmonid fish which had been force-fed, regurgitated food items contaminated with fenitrothion. Such behaviours would tend to reduce any impacts fenitrothion contamination of water or food may have directly on fish.

Fish may feed advantageously on the increased number of food items in the water column immediately after operational forestry spraying. Such opportunistic feeding apparently has no immediate, negative impacts on fish.

Studies attempting to examine fish population effects from food declines after spraying have been flawed. The primary shortcoming has been the inability to control fish immigration and emigration, which affect

population measurements. Total fish biomass within streams in operational forestry spray blocks has occasionally appeared to have been reduced, although not significantly. In a controlled stream-injection experiment which introduced concentrations of fenitrothion 10 to 100 times greater than those observed after operational sprays, and which caused massive invertebrate mortality, the fish biomass was reduced to 50% of control within one month of treatment. This effect was not due to fish mortality.

Information on the effects of operational fenitrothion spraying on fish growth is equivocal and difficult to interpret due to habitat variability. The extent to which spray-induced changes in food availability may affect fish populations remains a question. If such changes occur, however, the magnitude of measured invertebrate perturbations, which would expectedly be larger than those in fish, suggests they would be small.

4.1 INTRODUCTION

4.1.1 Overview and Objectives

Fenitrothion is an organophosphorus insecticide used worldwide for the control of insects important to agriculture, forestry, and human health (NRCC 1975). In Canada, fenitrothion is used to reduce damage caused by the spruce budworm (*Choristoneura fumiferana*) and other pests to coniferous forests. Fenitrothion has been used in most provinces and has been sprayed extensively in New Brunswick since the late 1960s (Symons 1977a).

In the 1980s, some provinces opted not to control forest defoliation, to spray less, or to switch to more environmentally acceptable treatments. At present, New Brunswick and Newfoundland are the only provinces which annually undertake large-scale aerial spraying of forests with fenitrothion.

A large body of literature (>400 references), in a variety of formats, deals with fenitrothion effects in aquatic systems. The aims of this review are to provide a background for understanding the work done on this subject, to summarize the relevant literature, and to discuss the ecological consequences of fenitrothion spraying, highlighting areas of concern.

4.1.2 Background and Definitions

Most work on the ecological effects of fenitrothion on aquatic fauna has been done in lotic (flowing water) systems. Little work has been done in lentic (standing water) systems. All monitoring work (defined as studies planned and carried out around operational spray programs) has taken place in lotic systems. Experimental work, on the other hand, does not involve the operational spray program; and the investigators usually simulate a dose by direct injection, ground-level spraying, or special aerial application. Such studies have been carried out in both lotic and lentic systems.

An important limitation to interpreting the results from about half of those studies is the lack of measurement of the fenitrothion concentration to which the aquatic organisms were exposed. The variability of deposit from aerial emissions and the variability of the actual concentrations obtained from experimental injections make it imperative that some measurement be made of fenitrothion amounts actually impinging on a given water body. The dosage rate is not a reliable indicator of the expected concentrations in water. Without a reliable measurement of exposure concentrations, it is impossible to determine whether varying responses of aquatic organisms may be due to differences in fenitrothion exposure or in the susceptibility of the aquatic fauna, or to their habitats.

4.1.3 Considerations for Interpretation of Results

Much of the ecological work done on fenitrothion effects on aquatic invertebrates suffers from one or more basic flaws in experimental design. Those shortcomings detract from the confidence one can place in the reported results and are mainly due to the difficulty of adequately quantifying benthic populations (Allen 1984; Elliot 1977) and to time and resource constraints faced by investigators (Symons 1977b). Allen (1984) provides an excellent review of those problems. A brief discussion is presented here to give a perspective on the interpretation of results.

Precision refers to the degree of similarity between repeated measures of the same attribute. Standard deviation, standard error, and 95% confidence limits are commonly used measures of precision. These precision measurements say nothing about the accuracy (a measure of how close the measured value is to the true value) of a measurement.

Therefore, one can have exceedingly precise measurements which are inaccurate.

Proper experimental design is necessary so that confidence may be placed in results obtained from any scientific inquiry. Impact studies, which evaluate the effects of a given perturbation, are no exception. Possible designs for different situations and potential treatments of results are well summarized and discussed by Green (1979). The most important considerations are controls in space and time, replication of treatments, and replication of random sampling within treatments. An effect can only be demonstrated in comparison to a control. Natural variation, dynamics, chance events, and sampling error provide many opportunities for spurious results to look like treatment effects.

The unfortunate reality of most aquatic impact studies is that treatments are rarely replicated and controls are less often replicated than treatments. Lack of real replication in treatments and controls leads to a situation where the replicates referred to in most works are really "pseudoreplicates" (Hurlbert 1984). The replicates are sub-samples from the same treatment and should not be used to make inferences attributing differences between areas to treatment or impact. There are never true controls, but quite often something that is better described as a reference site. Logistical constraints and the labour-intensive nature of environmental sampling are the main factors forcing that situation on investigators. That problem is common to a major proportion of field studies in ecology.

Most fenitrothion spray impact assessments have not been conclusive due to some of the above factors. Authors who realized the limitations of their work have shied away from statistical conclusions and used graphical or tabular presentations of results. Many of the statistical analyses which have been done were inappropriate, and the conclusions (e.g. no effects, which is potentially a type II error) were predetermined by the large variance and low sample number. Sometimes the analysis itself has not been valid because the assumption of replicated treatments has not been met. It should also be appreciated that, while individual studies may have been difficult to interpret, there now exists a relatively large database on the aquatic effects of fenitrothion from which inferences can be made on impact trends (see section 4.9).

4.1.4 Aquatic Buffer Zones

The Atlantic Provinces which continue to use fenitrothion in forest spray programs have implemented protective, no-spray buffer zones for certain water bodies. Buffer zones for fenitrothion spraying were first implemented in 1978 (Sexsmith 1987). Because much of the aquatic impact assessment was done before then, it might be assumed that the frequency of the measured impacts was greater than that which occurs today. Such an assumption is simplistic and has limitations.

In the first place, all of the large aircraft spraying in New Brunswick (in recent years, the largest portion of the program) and all of the spraying in Newfoundland has the requirement for 400-m buffer zones around lakes greater than 40 ha and along "major" rivers (Cahill 1987, Sexsmith 1987). Most of the studies have been conducted on much smaller water courses and, therefore, the major portion of fenitrothion spraying would not be subject to regulatory buffer zone constraints. According to the observations of regulatory inspectors, there is some effort on the part of spray pilots to avoid direct spraying of smaller water courses as a matter of good practice (K. Browne, New Brunswick Department of Municipal Affairs and Environment, pers.comm.). There is no indication of the effectiveness of such voluntary efforts, and no reason to suspect that they were not being made at the time that most aquatic impact data were being collected. However, an increased sensitivity on the part of spray applicators to environmental concerns in the recent past is generally acknowledged.

Secondly, although the portions of the New Brunswick spray program conducted by smaller agricultural aircraft are subject to 65-m buffer zones for all visible water bodies (Sexsmith 1987), there is no quantitative information which indicates that these buffer zones are effective in reducing deposit on small bodies of water when they are observed.

Considering these factors one must presume that for small water bodies fate and effect information, generated at a time prior to regulatory implementation of buffer zones, is probably indicative of what is occurring today.

4.2 CHEMICAL ANALYSIS

Fenitrothion is easily extracted from aqueous samples using an organic solvent such as hexane or ethyl acetate (Grift and Lockhart 1974; Ripley et al.

1974), and solid extraction using Amberlite XAD resins has been done successfully (Berkane et al. 1977; Mallet et al. 1978; Volpé and Mallet 1979; and Sergeant and Zitko 1979). One advantage of using XAD resins over solvent extraction is that fenitrothion may remain stable in the columns for several weeks. The use of Amberlite XAD-2 resin to recover fenitrothion and some degradation products (fenitrooxon, aminofenitrothion) from water has been compared with extraction by chloroform (Mallet et al. 1978). Both methods were found to be highly reliable for all three chemicals with relative error near 5% at 10 µg/L for fenitrothion.

For more difficult substrates such as foliage tissue and soils, water-miscible solvents such as acetonitrile or methanol are commonly used (Sergeant et al. 1979) but non-miscible solvents such as ethyl acetate, have been employed. Several clean-up methods have been developed (Grift and Lockhart 1974; Sergeant et al. 1979) most of which are tedious and time consuming. Haché et al. (1981) developed a fast clean-up method for difficult substrates using Amberlite XAD-7 columns applicable to the parent compound (5 µg/g), fenitrooxon (50 µg/g), and S-methyl fenitrothion (50 µg/g). A sample of soil or fish tissue may usually be processed in 2 h with excellent recoveries for fenitrothion at 0.05 µg/g.

The cleaned-up extract may be analysed by a variety of chromatographic methods. Zakrevsky and Mallet (1977) developed an *in situ* fluorometric method for the detection and quantitative analysis of fenitrothion and some degradation products (aminofenitrothion, 4-amino-3-methylphenol, fenitrooxon, and 3-methyl-4-nitrophenol). In principle, this method may be used to analyse 0.1 µg/L of a chemical extracted from water (Mallet et al. 1975; Mallet 1980). The most popular approach for the determination of fenitrothion and related products (fenitrooxon, aminofenitrothion, S-methylfenitrothion, formyl fenitrothion, and hydroxy-methylfenitrothion) is gas-liquid chromatography (GLC) (Forbes et al. 1975; Greenhalgh and Marshall 1976; Volpé and Mallet 1980). Another method gaining popularity is high-pressure liquid chromatography (HPLC) (Volpé and Mallet, 1981) which is less sensitive but is also useful for detecting the hydrolysis product 3-methyl-4-nitrophenol, and other similar by-products, without derivitization. Both the parent compound and its hydrolysis product may be analysed simultaneously by GLC after derivitization directly in water (Cassista and Mallet 1984).

4.3 FATE OF FENITROTHION IN AQUATIC SYSTEMS

4.3.1 Water

In laboratory studies, it has been demonstrated that fenitrothion is relatively stable to hydrolysis under acidic conditions, but the rate of hydrolysis increases rapidly with alkalinity (Sundaram 1973; Zitko and Cunningham 1974). The hydrolysis of fenitrothion and fenitrooxon as well as the formation of 3-methyl-4-nitrophenol proceeds only very slowly under environmental conditions. Table 4.1 identifies the known degradation products of fenitrothion. The data of Metcalfe et al. (1980) indicate that the volatilization rate of fenitrothion from solution may contribute significantly to the disappearance of fenitrothion in lake water. Malis and Muir (1984) determined that the half-life of ^{14}C labeled fenitrothion in pond water was less than 2 d. This is contrary to simulation modelling done by Marshall and Roberts (1977) which suggests a half-life of 93 d in this medium.

Greenhalgh et al. (1980) also observed that fenitrothion was readily hydrolyzed above pH 8 to 3-methyl-4-nitrophenol. Below pH 7, a second reaction involving dealkylation also took place forming desmethylfenitrothion. They determined that aminofenitrothion was formed only in natural lake water and not in distilled water. They attributed that to the action of bacteria likely to be present in natural lake water. An estimate of the half-life ranged from 50 d, in the dark under laboratory conditions, to 2 d in the field. That suggests that in the field, photolytic or microbial processes are the primary means of removing aminofenitrothion from natural water systems.

In previous studies of the photolysis of fenitrothion, Greenhalgh and Marshall (1976) and Miyamoto (1977) observed that carboxy fenitrothion and *p*-nitro-*m*-cresol were the predominant breakdown products. Mikami et al. (1985) studied the photodegradation of fenitrothion in water and on soil surfaces as well as its hydrolysis in water. That study indicated a rapid degradation in the presence

Table 4.1 Known degradation products of fenitrothion in environmental water.

Chemical	Source	Reference
Aminofenitrothion	Microorganisms (anaerobic)	Moody et al. 1978 Maguire and Hale 1980 Greenhalgh et al. 1980
Carboxyfenitrothion	Photolysis	Mikami et al. 1985
Desmethylamino-fenitrothion	Microorganisms (anaerobic)	Greenhalgh et al. 1980 Moody et al. 1978
Desmethylfenitrothion	Microorganisms	Greenhalgh et al. 1980
Fenitrooxon	Oxidation	Ohmae et al. 1981

Maguire and Hale (1980) observed that although chemical hydrolysis and volatilization from true solutions are both slow processes, volatilization of fenitrothion from surface slicks of its formulation sprayed on water is very fast ($t_{1/2} = 18$ min at 20°C). A field experiment used a stagnant pond sprayed with a water formulation containing 11% (w/w) fenitrothion at a dosage rate of 1500 g a.i./ha. Fenitrothion concentrations in water, suspended solids, and sediment fell below detectable levels 2d after the spray. Three-methyl-4-nitrophenol and aminofenitrothion, in very small quantities, were identified in water and sediment, respectively, but neither chemical was detected after 4 d.

of sunlight and the formation of many degradation products. A photodegradation pathway for fenitrothion in water and soil was suggested. One main oxidation route seems to be via the 3-methyl group which is converted to carboxyl. Earlier, Weinberger et al. (1982a) obtained somewhat similar results, finding the photolysis of fenitrothion generally unaffected by pH of waters. In all systems, 80% of the pesticide was photolyzed to polar derivatives within 6 h. Earlier, Moody et al. (1978) and Lakshminarayana and Bourque (1980) conducted similar, complementary studies. The persistence of fenitrothion seems to be somewhat greater in sea water (Kodama and Kuwatsuka 1980), although the reasons are not understood.

Moody et al. (1978) studied the fate of fenitrothion in stream water following aerial spraying (no indication of stream buffer zone) with an aqueous formulation of the insecticide at the rate of 420g a.i./ha. In stagnant water fenitrothion levels ranged from 770 µg/L at the surface after 1 h to 0.67 µg/L after approximately 100 h. They found traces of desmethylaminofenitrothion, S-methylfenitrothion (4.4 µg/L), and aminofenitrothion (0.05 µg/L).

Ohmae et al. (1981) studied the environmental behaviour of fenitrothion and its decomposition products after operational aerial application. The initial concentration of fenitrothion in water (38.2 µg/L) immediately after application was rapidly reduced although small amounts of parent fenitrothion (0.02 µg/L) and small amounts of 3-methyl-4-nitrophenol were detected after 49 d.

Eidt et al. (1984) studied the partitioning and short-term persistence of fenitrothion in New Brunswick headwater streams. They found that the difference between concentrations of fenitrothion at various depths and positions in the water column was not significant.

Thus, while laboratory studies (Greenhalgh et al. 1980) indicate that fenitrothion has a half-life of more than 50 d in water, field studies clearly show that only small amounts (limit of detection 0.01 µg/L) remain after a few days. Processes such as dilution, volatilization, photolysis, and chemical and microbial degradation all play an important role in its disappearance in nature.

4.3.2 Aquatic Plants

Fairchild and Eidt (1988) studied the impact of fenitrothion upon the aquatic organisms in a small bog pond in New Brunswick. They found that although residues of fenitrothion in water dropped below detection levels after only a few days, residues reappeared the next spring just before commencement of the spray program. That release of fenitrothion was suspected to come from biotic sinks, particularly moss.

Kikuchi et al. (1984) observed that three species of algae rapidly adsorbed fenitrothion which had been metabolized to the oxon, the desmethyl analog, or the corresponding phenol, depending on the alga.

Baarchers and Heitland (1986) studied the biodegradation of fenitrothion and fenitrooxon by the fungus *Trichoderma viride* which was observed to be capable of hydrolyzing fenitrothion and fenitrooxon in pure culture as well as co-metabolizing

the nitrophenol. Miyamoto (1977) had previously found that bacteria and some fungi converted the pesticide to aminofenitrothion.

As stated earlier, the method of choice for the analysis of fenitrothion involves extraction with organic solvents and analysis by gas-liquid chromatography. That method, however, will only yield information for those chemicals readily extractable and identifiable by comparison with a known standard. It does not answer the question as to what happens to the undetected material. For that reason, ¹⁴C labeled fenitrothion was used to study the fate of fenitrothion in shaded and unshaded ponds (Malis and Muir 1984). Initial concentrations of the insecticide in water were about 70 µg/L. Aquatic macrophytes, fish, and sediments absorbed fenitrothion and some insecticide vaporized into the air such that 90% of the radioactivity could be accounted for after 2 d post-treatment. The accountability was reduced to less than 30% after 21 d. Fisher (1985) evaluated ¹⁴C labeled fenitrothion in aquatic microcosms and found the uptake of fenitrothion by organisms such as algae and snails was greater at pH 8.3 than at pH 6.7. That observation was probably related to the more rapid hydrolysis of fenitrothion at higher pH.

Weinberger et al. (1982b) studied the fate of formulated ¹⁴C labeled fenitrothion in various aquatic microcosms. They found that in all microcosms, the aquatic macrophytes and the algal microphytes accumulated fenitrothion to a greater degree in the light than in the dark. Weinberger et al. (1983) studied the effect that spray formulation additives had on the residue dynamics of fenitrothion. They observed that Dowanol, when compared to Arotex, led to a two-fold increased uptake of ¹⁴C labeled fenitrothion by water microphytes. The nature of those additives to some extent determined the amount of uptake and the array of derivatives measured in the algae.

4.3.3 Sediments

According to Baarchers et al. (1983) and Eidt et al. (1984), adsorption of pesticides and their degradation products on suspended particles, soils and sediments play a major role in their environmental fate. They also observed that fenitrothion and 3-methyl-4-nitrophenol (the major degradation product in that context) are adsorbed on sediment and soils in amounts that rise with increasing organic matter content.

4.4 EXPOSURE IN THE AQUATIC ENVIRONMENT

4.4.1 Factors Affecting Exposure

The formulation and rate of application of the emitted aerial spray influences potential exposure. The presence of oils and other substances in the spray formulation can alter the distribution of fenitrothion on contact with water. Oils can be concentrated in surface slicks, while other adjuvants such as emulsifiers or surfactants could speed up dispersion in the water column. Higher dosage rates increase the chances of significant amounts of fenitrothion reaching water. Spray cloud behaviour is influenced by application technology, topography, and meteorological conditions. The deposit actually reaching open water is influenced not only by all these factors, but also by the presence of buffer zones and the density of the forest canopy shielding the water from the spray.

Once fenitrothion is in the water, the duration and magnitude of the peak concentration is influenced by factors such as the size of the water body, by the rate of exchange, and by the physical and chemical properties of the water. In large water bodies, deposited pesticide is dispersed and diluted more effectively than in small ones. The rate of water exchange in the system also influences the duration of exposure. Lotic systems, with constant flushing, have more transitory exposure than do lentic systems although exposure is more widespread in flowing systems. The rate of degradation of fenitrothion is influenced by water chemistry and light penetration, which affect the rates of hydrolysis and photolysis. Dark, peaty, acidic water has lower potential for degradation than clear, neutral-to-basic water.

Localized high concentrations of fenitrothion may be found in surface slicks, pools in streams, and small ponds. Surface slicks may prevent the dispersion of fenitrothion. Such slicks may also be moved by wind, creating higher than expected concentrations on exposed shorelines. Pools in streams can receive a greater input of insecticide if they are large enough to have reduced canopy cover. Such pools have the potential for acting as slow-release reservoirs in streams which could prolong the duration of exposure in riffle areas downstream. Small ponds are difficult to avoid by applicators trying to respect buffer zones, as such water bodies are often not marked on maps and can be difficult to see from the air. Small ponds have limited capacity for dilution, limited flushing, and often limited forest-

canopy protection. They have been identified as areas that are potentially at risk from forest spraying (NRCC 1982).

4.5 AQUATIC RESIDUES DETECTED BY MONITORING PROGRAMS

Maximum recorded aqueous concentrations of fenitrothion in lotic systems after operational spraying (Table 4.2) usually occur within 2 h of the start of spraying and range from 1.30 $\mu\text{g/L}$ (Mallet and Cassista 1984) to 127 $\mu\text{g/L}$ (Morin et al. 1986). The median of all the maximum concentrations measured after operational spraying was less than 10.0 $\mu\text{g/L}$. Many of the higher maximum concentrations recorded were: in surface film samples (Morin et al. 1983, 1986); in samples from streams which were not protected by buffer zones and had sparse forest-canopy cover (Coady 1978); or were associated with other circumstances (Gaboury et al. 1981; Morin et al. 1986). When studies were carried out such that mean maximum concentrations could be calculated for a particular spray application (Table 4.2), concentrations ranged from 0.59 $\mu\text{g/L}$ (Sundaram et al. 1984) to 18.4 $\mu\text{g/L}$ (Flannagan 1973). For all studies reviewed, the mean maximum concentration was 9.6 $\mu\text{g/L}$ (Table 4.5). Fenitrothion concentrations usually declined to less than 1.0 $\mu\text{g/L}$ within 24-48 h, and less than 0.5 $\mu\text{g/L}$ within 1-6 d (Gaboury et al. 1981).

Maximum recorded aqueous concentrations of fenitrothion in lentic systems after operational spraying usually occur within 2 h after the start of spraying and range from 0.38 $\mu\text{g/L}$ (Major and Dostie 1983) to 1114 $\mu\text{g/L}$ (Morin et al. 1986) (Table 4.3). Median peak concentration was 12.2 $\mu\text{g/L}$ for the 16 studies recorded (Table 4.5). The mean and median half-life of fenitrothion in lentic water was 18.9 h and less than 8.5 h respectively. Morin et al. (1986) found that the high concentration of 543 $\mu\text{g/L}$ in a pond was reduced by 95% within 3.3 h after spraying.

Recorded concentrations of fenitrothion in sediment after operational spraying are less than 0.5 $\mu\text{g/g}$ (Table 4.4). Half-life in sediment has not been determined.

Residue concentrations detected in representative biota, other than fish, are presented in Table 4.6. Fenitrothion residues found in biota after operational spraying range in concentration from 0.014 $\mu\text{g/g}$ in

zooplankton (Lakshminarayana and Bourque 1980) to 4.19 µg/g in duckweed (Moody et al. 1978). Generalizations about concentrations in biota are of limited value because each organism has a fairly unique capacity to accumulate fenitrothion, determined by its behaviour and physiology. Bivalve molluscs are of particular concern. Several studies in New Brunswick (reported in Varty 1980) have indicated that oysters (*Crassostrea virginica*), clams (*Mya* sp.), and mussels (*Mytilus edulis*) were contaminated with fenitrothion residues after the end of spraying in that year. In the laboratory, however, residues in bivalve molluscs have been observed to accumulate slowly and to be excreted rapidly, providing a low accumulation coefficient. The accumulation coefficients (uptake/excretion) range from nine to 130, depending upon the test species (McLeese et al. 1979).

4.6 TOXICITY OF FENITROTHION TO AQUATIC INVERTEBRATES

4.6.1 Mode of Action

"The basic mechanism responsible for the toxicity of organophosphorus insecticides is the inhibition of acetylcholinesterase of the nervous system, with the consequent accumulation of excessive levels of acetylcholine" (NRCC 1975). Fenitrothion affects the nervous system and sublethal effects can manifest themselves as nervous system changes or behavioural changes.

4.6.2 Aquatic Invertebrate Sensitivity

Aquatic invertebrates exhibit a wide range in sensitivity to fenitrothion, some 24 h LC50s being as low as 0.4 µg/L for *Culex tarsalis* (Diptera) and 2.0 µg/L for *Acronuria* sp. (Plecoptera) (Wildish and Phillips 1972; Flannagan 1973). Other invertebrates are much more tolerant, e.g. 24 h LC50 of 40 000 µg/L for *Eriocera spinosa* (Diptera) and 72 h LC50 of 20 000 µg/L for *Brachycentrus numerosus* (Trichoptera) (Wildish and Phillips 1972; Symons and Metcalfe 1978).

Summaries and discussions of specific toxicities for a wide range of invertebrate taxa are available (NRCC 1975; Symons 1977a; Johnson and Findlay 1980; Woodward and Mauck 1980; Gaboury et al. 1981; Sanders et al. 1983; Poirier and Surgeoner 1986).

Sensitivity of organisms under laboratory conditions is, however, not as important as ecological sensitivity (Ahern and Leclerc 1981; Poirier 1986). Many organisms avoid (actively and passively) fenitrothion in water through burrowing, habitat selection, or other behaviours. For many organisms, therefore, the pesticide concentration present in water may not be an accurate measure of the actual concentration to which the organism is exposed.

Specific sensitivities, as measured by conventional toxicological procedures (i.e. 24-96 h LC50s), also have limited predictive capability for the short-pulsed exposure fenitrothion organisms receive in the field (Poirier 1986). Short pulses of high concentration may have more or less severe consequences than longer exposures to a lower concentration, even though the products of concentration and time may be equivalent (Muirhead-Thomson 1973).

4.7 EFFECTS ON INVERTEBRATE DRIFT

4.7.1 Characterizing Drift

Drift refers broadly to the downstream transport, by current, of lotic benthic animals normally living on or in bottom substrates (Wiley and Kohler 1984). This phenomenon has been extensively studied, with emphasis on factors influencing drift and its diel periodicity. Drift has been classified into three categories: catastrophic or pulsed; behavioural or periodic; and constant or continuous. These categories are based on the temporal dimension of drift and are not thought to be exclusive. Drift is probably better described as "a composite phenomenon that is the result of a myriad of independent, simultaneous, and to some extent, unknown causes" (Wiley and Kohler 1984).

Drift provides aquatic insects with an immediate and energetically inexpensive way to escape or disperse. Active entry into the water column is a common behavioural response of aquatic insects to stress stimuli such as predation, competition, random contact, or response to physical or chemical irritants (Scherer and McNichol 1986; Wiley and Kohler 1984). Passive entry into the drift is the result of accidental or unintentional release of the substrate and subsequent erosion into the water column.

Scherer and McNichol (1986) showed that fenitrothion exposure in a laboratory stream system caused *Acronuria lycorias* to increase in drift at the

Table 4.2 Concentration of fenitrothion residues in lotic water after operational aerial spraying.

Location	Maximum concentration		Time post spray (h)	Estimated Half-life (h)	Reference
	Peak ($\mu\text{g/L}$)	Mean (Median) ($\mu\text{g/L}$)			
PQ	0.356	-	<24	-	Dostie and Parent 1986
(>20 km downstream)					
NB	1.30	-	-	-	Mallet and Cassista 1984
MB	1.32	-	1	<10	Moody et al. 1978
(fast water)					
NB	1.33	-	0.66	2	Eidt and Sundaram 1975
PQ	1.74	-	1.5	<4.5	Holmes 1979
NB	1.84	0.59 (0.21)	1.0	3-6	Sundaram et al. 1984
PQ	2.63	-	1	-	Mathieu et al. 1979
PQ	3.79	-	1	<3	Mamarbachi et al. 1987
NB	3.8	-	3.5	8	Eidt 1975a
NB	4.84	-	1.0	3.5	Peterson and Zitko 1974
NB	5.25	-	0.66	8.0	Eidt and Sundaram 1975
PQ	5.94	(4.73)	1-2	-	Dostie et al. 1985
PQ	6.03	-	6	-	Dostie et al. 1987
(1.5-3.0 km downstream)					
NB	6.38	-	0.91	<8.0	Eidt and Sundaram 1975
ME	6.62	3.46 (3.26)	24	-	Marancik 1976a
ON	6.7	-	0.5	84	Sundaram 1974a
NB	7.4	-	2	-	Banks 1973
PQ	7.43	-	5	<24	Mamarbachi et al. 1985
PQ	8.71	-	1.5	<3	Mamarbachi et al. 1987
PQ	8.72	5.28 (6.42)	2	-	St. Jean and Mamarbachi 1987
PQ	9.08	-	-	24	Major and Dostie 1983
PQ	10.56	-	1	-	Foisy et al. 1977
PQ	12.8	-	2	<6	Sarrazin 1977
PQ	13.4	8.7	0.75	<6	Marotte 1981
PQ	14.2	(2.53)	-	-	Morin et al. 1983
Scotland	18.0	9.4 (5.1)	1-2	<12	Morrison and Wells 1981
NB	20.0	most <10	-	-	Mallet and Volpé 1982
NB	21.0	5.34 (3.75)	1-2	-	Eidt 1977
NB	28	~18.25 (17.5)	1-24	-	Montreal Engineering Co. 1978
WA	32	8.2 (3.5)	-	-	Tracy and Purvis 1976
NB	36	-	~4	-	Lord et al. 1978
	42	-	~6	>16	
Japan	38.2	-	<0.5	<2	Ohmae et al. 1981
Scotland	48.0	15.4 (8.5)	<4	<12	Wells et al. 1978a
NF	49	-	0.08	<6	Coady 1978
PQ	55.1	8.13 (1.16)	-	-	Morin et al. 1983
NB	57.0	14.1 (5.6)	0.83-3.0	-	Hall in Eidt and Sundaram 1975
MB	64.0	18.4 (4.1)	1-6	-	Flannagan 1973
MB	75.5	-	17	>24	Lockhart et al. 1973
PQ	127.0	9.45 (2.84)	1	-	Morin et al. 1986

Table 4.3 Concentration of fenitrothion residues in lentic water after operational aerial spraying.

Location	Maximum concentration		Time post spray (h)	Estimated Half-life (h)	Reference
	Peak ($\mu\text{g/L}$)	Mean (Median) ($\mu\text{g/L}$)			
PQ	0.38	-	-	24	Major and Dostie 1983
NB	0.9	-	8.5	-	Lakshminarayana and Bourque 1980
PQ	<1.0	-	-	-	Sarrazin 1976
PQ	3.43	-	72	-	Kingsbury 1973
NB	4.9	3.53	10	<24	Ernst et al. 1980
	1.4	1.03	24	<72	
PQ	8.04	-	6	-	Dostie et al. 1985
NB	11.8	-	-	-	Mallet and Volpé 1982
PQ	12.69	-	1	-	Mathieu et al 1979
ON	25.5	19.5	0-0.05	6-60	Sundaram 1974a
PQ	30.1	7.29	-	-	Gaboury 1980
PQ	51.56	-	1	<7	Holmes et al. 1984
PQ	202	-	1.5	<5	Mamarbachi et al. 1985
PQ	543	(8.06)	1	<3	Morin et al. 1983
MB	701.0 (slow water)	-	1	<10% at 10h	Moody et al. 1978
PQ	1114.0	53.57 (5.82)	1	-	Morin et al. 1986

Table 4.4 Concentration of fenitrothion residues in sediments after operational aerial spraying.

Loc.	Maximum concentration ($\mu\text{g/g}$)	Mean (Median) ($\mu\text{g/g}$)	Time post spray (h)	Reference
MB	0	0	anytime	Flannagan 1973
Scotland	0	0	anytime	Morrison and Wells 1981
NB	>0.01	-	-	Mallet and Cassista 1984
NB	0.05	-	-	Wells et al. 1979
NB	0.1	-	10	Ernst et al. 1980
PQ	0.461	-	6	Mathieu and Provancher 1978

Table 4.5 Summary of concentrations of fenitrothion residues detected in water and sediment after operational aerial spraying.

Location	Maximum conc.			Mean max. conc.			Median max. conc.			Half-life (h)		
	n	mean	s	n	mean	s	n	mean	s	n	mean	s
		(median)			(median)			(median)			(median)	
lotic water ($\mu\text{g/L}$)	39	22.0	26.4 (9.08)	13	9.6	5.5 (8.7)	14	4.9	4.2 (3.9)	21	12.	17.8 (<6)
lentic water ($\mu\text{g/L}$)	16	170	328 (12.2)	5	17.0	21.6 (7.29)	2	6.9	1.6 (-)	8	18.9	23.0 (<8.5)
sediments ($\mu\text{g/L}$)	7	0.30	0.37	—	—	—	—	—	—	1	<72	—

n = number of records used in calculation

s = standard deviation

Table 4.6 Concentration of fenitrothion residues in biota other than fish after operational spraying.

Biota	Concentration $\mu\text{g/g}$	Time post spray	Estimated Half-life (h)	Reference
Zooplankton	0.014	<2 h	—	Lakshminarayana and Bourque 1980
Phytoplankton	0.032	<2 h	—	Lakshminarayana and Bourque 1980
Plankton	0.422	24 h	—	Mathieu and Provancher 1978
Anodonta sp.	1.4	24 h	—	McLeese et al. 1980 in Varty 1980
Clams	<0.005	—	—	Wells et al. 1979
Mussels	<0.005	—	—	Wells et al. 1979
Oysters	<0.005	—	—	Wells et al. 1979
Clams	0.026	72 h	—	Kingsbury 1973
Mollusk Shellfish	0.01-0.40	—	—	Mallet 1978
Oyster (<i>C. virginica</i>)	0.03-3.7	10 d	—	Lord et al. 1978
Mussels (<i>M. edulis</i>)	0.02-0.19	10 d	—	Lord et al. 1978
Aquatic plants	0.012 (median)	—	—	Montreal Engineering Co. 1982
Aquatic moss	0.139	9 h	—	Morrison and Wells 1981
Hornwort (<i>C. demersum</i>)	0.15	70 h	—	Moody et al. 1978
Rush (<i>B. umbellatus</i>)	0.35	70 h	—	Moody et al. 1978
Duckweed (<i>L. minor</i>)	4.19	10 h	<24	Moody et al. 1978

(continued)

Table 4.6 (continued)

<i>Biota</i>	<i>Concentration μg/g</i>	<i>Time post spray</i>	<i>Estimated Half-life (h)</i>	<i>Reference</i>
Invertebrates	0.101	33 h	—	Mamarbachi et al. 1987
Pooled Invertebrates	0.27 (median)	—	—	Monenco 1982
Mayflies	3.2	—	—	Penney 1971c
Crustaceans	0	anytime	—	Wells et al. 1979
Crayfish (<i>O. virilis</i>)	1.37	19 d	—	Leonhard 1974
<i>Bufo americanus</i>	0.123	1.5 h	<3	Mamarbachi et al. 1987

onset of the first period of darkness following exposure. Their results suggested that drift was due to increased movement by the insects after fenitrothion exposure.

4.7.2 Measuring Drift

Drift is measured by intercepting organisms in the water column with a net. A measurement can then be made of the number of organisms per volume of flow through the net. Nets are set for various time intervals usually ranging from 10 min to 24 h. As a net fills, its collecting efficiency declines. A series of shorter measures provides a more accurate measurement of drift than a net emptied less frequently. A net emptied only once in 24 h may also completely miss important changes in drift patterns between day and night. Drift is non-uniform from the surface to the substrate. To obtain standard samples, it is necessary to extend a net from above the water surface to the stream bottom. If the entire column cannot be sampled, the bottom flow is more important because much of the drift occurs in that area (Eidt 1981).

The complexity of drift reactions makes it important to establish the proportion of dead insects in the drift after forest spraying. An increase in the dead portion can be attributed to the intervention of a lethal agent. An increase in total drift without a measure

of mortality may simultaneously reflect mortality, active drift in response to the chemical (irritant effect), and passive drift as a result of increased activity (sublethal behavioural effect). The relative contribution of each factor is very difficult to determine in the field and all may be important losses of an energy source to a particular section of stream.

Drift studies which do not provide adequate controls are at risk to a wide range of possible interference factors. In addition to those described above, drift is also influenced by the phase of the moon and by current velocity. Differences in canopy cover (affecting incident moonlight) or in rainfall (affecting current) between areas may cause differences in drift patterns. Changes in drift patterns are often difficult to understand and often lend themselves to more than one interpretation.

4.7.3 Effects of Fenitrothion on Drift

The drift studies reviewed have been summarized in three groups: field monitoring without insecticide residue analysis; field monitoring with insecticide residue analysis; and experimental field studies. Dosage rates in these studies range from 140-420 g a.i./ha. Measured insecticide residues in water range from 0.25-64 μg/L. Experimental studies have generally employed higher dosage rates and higher

fenitrothion concentrations than would be expected after operational spraying.

Field monitoring studies without fenitrothion residue analysis are presented in Table 4.7. Only four of 12 entries in Table 4.7 indicate no effects on invertebrate drift. Two of those studies were at a dosage rate of 2×140 g a.i./ha (Penney 1971b; Dimond and Malcolm 1971) and two were at 2×210 g a.i./ha (Riche and Coady 1969; Montreal Engineering Co. 1976). Large increases in drift were noted at the highest dosage rates of 280–420 g a.i./ha (Vincent 1969; Welch, in Symons 1977a). Invertebrate groups were Ephemeroptera (Riche and Coady 1969; Rabeni and Gibbs 1976b), Ephemeroptera and Plecoptera (Montreal Engineering Co. 1976, 1978), and Ephemeroptera, Plecoptera, and Hemiptera (Carrow 1974). When drift effects were observed the magnitude of the effects correlated well with the spray dosage rate.

Field monitoring studies which measured fenitrothion residues are identified in Table 4.8. Studies are presented in ascending order of fenitrothion residue concentrations. Only six of 22 studies showed no effects on drift. Five of the six "no-effects" studies occurred at fenitrothion concentrations of less than $4.0 \mu\text{g/L}$ (Eidt 1975b, 1977; Holmes 1979, 1982a). The remaining "no-effects" study (Banks and Hall 1972) was exceptional in that it was conducted during late summer spruce budworm adulticiding (the others being done during late spring larviciding) and a relatively high fenitrothion concentration ($7\text{--}14 \mu\text{g/L}$) was measured. Only two studies recorded effects at concentrations less than $4.0 \mu\text{g/L}$ (Eidt 1975a; Holmes 1982a). Ephemeroptera and Plecoptera again were mentioned as the most sensitive groups (Eidt 1975a, 1975b, 1977; Peterson and Zitko 1974; Tracy and Purvis 1976; Lord et al. 1978). Holmes (1982a), Flannagan (1973), and Lord et al. (1978) also indicated that Diptera were affected.

The magnitude of drift correlates with the measured fenitrothion concentration in water with few effects occurring below a concentration of $4.0 \mu\text{g/L}$. The higher fenitrothion concentrations ($18\text{--}64 \mu\text{g/L}$) were associated with some of the larger drift events. Mortality in the drift does not show as clear a relationship. Heavy mortality was noted at $5.25 \mu\text{g/L}$ (Eidt 1977) and no mortality observed at $31.0 \mu\text{g/L}$ (Tracy and Purvis 1976). The lack of mortality in caged insects exposed to peak concentration of $18 \mu\text{g/L}$ indicates that such concentration may cause displacement of organisms but not mortality (Morrison and Wells 1981).

Most field experiments which measured fenitrothion-induced impact on drift employed dosage rates which were higher than would be expected in operational spraying (Table 4.9). The lowest dosage rate studied was 280 g a.i./ha ; and the measured residues ranged from $4.06 \mu\text{g/L}$ (Holmes 1982a) to $500 \mu\text{g/L}$ (Peterson and Zitko 1974). Patterns of drift in response to fenitrothion exposure were taxa specific (Yasuno et al. 1981; Hasegawa et al. 1982). Eidt (1981) also observed that organisms within the substrate sustained lower mortality than the drifting organisms in a stream that had a several-fold increase in drift due to fenitrothion exposure. One study emphasized the importance of having drift-net samples of the entire water column by showing affected organisms to be more abundant near the bottom than at the stream surface (Eidt and Weaver 1984a). Most peaks in drift occurred within the first 12 h after spraying (Table 4.9) and effects on drift generally lasted less than 24 h (Eidt 1977; Morrison and Wells 1981).

4.8 EFFECTS ON BENTHIC POPULATIONS

4.8.1 Characteristics of Benthic Populations

Benthic invertebrate populations are usually contagiously distributed (clumped) in response to patchy environmental factors and behaviour of the organisms. The usual result is a large variance in a set of benthic invertebrate samples (Elliot 1977). The large variance associated with population means makes comparisons among means for few samples difficult (Section 4.8.2). Populations are also subject to major temporal fluctuations due to insect emergence and oviposition.

Experimental controls are imperative if natural fluctuations are to be separated from chemically-induced impacts. Fenitrothion spraying (May to June) coincides with a very active time of insect emergence in most aquatic systems. If emergence is not measured or controls are not included in the experimental design, population declines due to natural causes cannot be separated from declines due to pesticide spraying.

4.8.2 Measurement of Benthic Populations

The characteristics of benthic populations pose a major problem for quantitative sampling. Needham and Usinger (1956) investigated the precision of

Table 4.7 Effects of fenitrothion on aquatic invertebrate drift after aerial spraying, based on insecticide dosage rate (with no corollary insecticide residue analysis).

<i>Dosage rate g a.i./ha</i>	<i>Measured effect</i>	<i>Reference</i>
2 x 140	no effects attributable to spraying	Penney 1971b
2 x 140	drift alive no major effects (24 h samples)	Dimond and Malcolm 1971
2 x 140	increased drift up to 48 h, mainly Plecoptera, Ephemeroptera, and Hemiptera	Carrow 1974
1 x 210	drift increased in two of three streams, about one-third dead, mostly Ephemeroptera	Rabeni and Gibbs 1976b
1 x 210	slight increase in drift	Penney 1971b
2 x 210	no effects attributable to spraying	Riche and Coady 1969
2 x 210	no effects attributable to spraying	Montreal Engineering Co. 1976
2 x 210	significant effect on both total and dead drift after second spray, Ephemeroptera, Plecoptera significant increase	Montreal Engineering Co. 1976
2 x 210	substantial increase in Ephemeroptera drift after first spray (no adequate control)	Riche and Coady 1969
2 x 210	increase in both dead and living drift, not significant, dead percentage high in control Ephemeroptera and Plecoptera most sensitive	Montreal Engineering Co. 1978
1 x 280	large increase in drift	Welch in Symons 1977a
1 x 420	large increase in drift (one stream studied)	Vincent in Dimond and Malcolm 1971

Table 4.8 Effects of fenitrothion on aquatic invertebrate drift after aerial spraying (with corollary insecticide residue analysis).

<i>Residue level (Dosage rate g a.i./ha)</i>	<i>Measured effect</i>	<i>Reference</i>
0.25 µg/L (2 x 210)	no effects on drift; possible increase in Simuliidae and Chironomidae after second spray (no control)	Holmes 1982a
1.33 µg/L (210)	drift slightly above background levels Plecoptera and Ephemeroptera most affected	Eidt 1975a
1.74 µg/L (210)	no significant effect on drift; slight increase in Ephemeroptera drift	Holmes 1979
1.84 µg/L (2 x 210)	Ephemeroptera drifted for longer time	Holmes 1982a
2.0 µg/L (2 x 210)	no effect on drift	Eidt 1977
3.13 µg/L (2 x 210)	no effect on drift	Eidt 1977
3.8 µg/L (210)	drift not affected, three stream study; one stream possible increase in Ephemeroptera and Plecoptera	Eidt 1975b
4.84 µg/L (210)	increased Plecoptera in drift, suggestion of increase in Ephemeroptera drift; no evidence of effects on other groups	Peterson and Zitko 1974
5.25 µg/L (210)	drift high with large dead component; Plecoptera and Ephemeroptera most affected	Eidt 1975a
5.25 µg/L (2 x 210)	five-fold increase in drift, mainly dead insects, Ephemeroptera and Plecoptera drift returned to normal in less than 12 h	Eidt 1977
6.38 µg/L (210)	drift increased with larger dead component Plecoptera and Ephemeroptera most affected	Eidt 1975a
3.0-6.7 µg/L (2 x 140)	slight increase in drift, subtle but consistent; no dead drift noted; drift had more Ephemeroptera, less Diptera	Tracy and Purvis 1976

(continued)

Table 4.8 (continued)

<i>Residue level (Dosage rate g a.i./ha)</i>	<i>Measured effect</i>	<i>Reference</i>
11.2 µg/L (2 x 140)	increased drift	Hall and Eaton 1973
7-14 µg/L (2 x 140)	no significant change in drift behaviour (late summer adulticide application)	Banks and Hall 1972
7.4-14.3 µg/L (2 x 140)	substantial increase in drift after spraying	Banks and Hall 1972
18 µg/L (300)	drift increased markedly 12-16 h after spray; caged insects remained alive 5 d; suggests drift displaced not dead	Morrison and Wells 1981
31.0 µg/L (210)	slight increase in drift, subtle but consis- tent; no dead drift noted; drift had more Ephemeroptera, less Diptera	Tracy and Purvis 1976
38 µg/L (2 x 210)	increase in drift; most moribund or dead; Ephemeroptera then Diptera most affected	Lord et al. 1978
42 µg/L (2 x 280)	increase in drift; most moribund or dead; Ephemeroptera and Plecoptera most affected	Lord et al. 1978
4.2-48 µg/L (300)	sharp increase in drift within a few hours; decreased rapidly as concentrations de- creased; wide range of species affected	Wells et al. 1978a
39-49 µg/L (2 x 210)	marked alterations in drift patterns	Coady 1978
64 µg/L (280)	large increase in drift of the most common animals immediately after spraying	Flannagan 1973

sampling stream riffles and concluded that 73 Surber samples would be necessary for a precise estimate of mean total numbers and 194 samples for mean wet weight. Resh (1979) reanalyzed some of his own data, as well as earlier data, and concluded that "purely quantitative routine sampling in streams to determine weights and numerical data is impractical" (in Allen 1984). All students of stream ecology labour under the same burden, and investigators of fenitrothion effects are no exception.

What that means in practical terms is that measurements with small sample numbers can only detect gross changes; and the number of samples required for even modest precision is beyond the time and labour constraints of almost all investigators. Allen (1984) provided a more detailed discussion and summarized precision of benthic sampling as follows: "modest precision (e.g. 95% CL $\pm$ 50% of the mean) likely requires at least 10 replicates, and high precision (e.g. 95% CL $\pm$ 15-25% of the mean) likely requires greater than 50 replicates." The most replicate samples routinely taken in any of the fenitrothion field studies reviewed was nine (Banks 1973; Rabeni and Gibbs 1976b; Tracy and Purvis 1976). While one study employed 20 replicate samples (Penney and MacDonald 1966), most studies had between three and six replicates.

4.8.3 Fenitrothion Effects on Benthos

The studies reviewed have been divided into two groups, monitoring (41 entries) and experimental (15 entries) and are further subdivided into those with insecticide residue analysis and those without. All of the field monitoring of fenitrothion spraying has been done in lotic systems (Tables 4.10 and 4.11). Experimental work has been carried out in lotic, lentic, and other special habitats (Table 4.12 and 4.13).

Seven of 41 monitoring studies (Tables 4.10 and 4.11) demonstrated fairly clear reductions in total benthos after spraying (Dimond and Malcolm 1971; Banks and Hall 1972; Elson et al. 1972; Hall and Eaton 1973; Carrow 1974; Buckner et al. 1976; Welch, in Symons 1977a; Buckner and McLeod 1977; Montreal Engineering Co. 1978; Holmes 1982a) or reductions in only one or two taxa (Penney 1971b; Rabeni and Gibbs 1976b; Holmes 1982a). Only one monitoring study (Coady 1978) had corollary residue analysis (39-49 $\mu\text{g/L}$) to accompany clear effects (70-80% reduction in benthos), although three others suggested effects at concentrations ranging from 1.84-14.3 $\mu\text{g/L}$ (Banks and Hall 1972; Hall and

Eaton 1973; Holmes 1982a). None of those studies employed large enough numbers of samples (> 10) for statistically reliable conclusions, and most were inadequately controlled.

Experimental studies in lotic systems at three dosage rates with no accompanying residue analysis (Table 4.12) demonstrated a dose-dependent response. At $2 \times 280 \text{ g a.i./ha}$, Kingsbury (1978a) observed no depletion of benthos. A single 560 g a.i./ha application reduced numbers and weight of benthos by two-thirds based on 20 Surber samples (Penney and MacDonald 1966). An estimated 1000 $\mu\text{g/L}$ treatment eliminated all but two taxa from a stream in Japan (Hasegawa et al. 1982). The 560 g a.i./ha treatment is twice the highest rates used in Canadian forest spray programs. A treatment of 1000 $\mu\text{g/L}$ is 100 times higher than the mean maximum concentration found in lotic water after forest spraying.

Holmes (1982a) found no reduction in standing crop in either of two streams sampled in fenitrothion-treated areas. However, in the stream with the lower fenitrothion concentration (4.06 $\mu\text{g/L}$), Ephemeroptera and Trichoptera were reduced; while paradoxically, in the stream with the higher concentration (21.45 $\mu\text{g/L}$), there was no reduction in any taxa. In a stream injection experiment (10-23 $\mu\text{g/L}$ for 3 h), benthos was reduced by about 50%. Most of the kill consisted of *Leuctra* spp. (Plecoptera), and recovery appeared complete in 50 d (Eidt 1981). Eidt (1978) demonstrated that organisms buried in the substrate sustained less mortality than those at the substrate surface. Benthos was reduced 80-90% by a high concentration injection (100-460 $\mu\text{g/L}$ for 1.3 h), with *Simulium* recovering to above pre-treatment levels and *Anisogammarus* being eliminated (Yasuno et al. 1981). Ahern and Leclerc (1981) used results from a series of sites to establish a lethal threshold to benthic communities of 20 $\mu\text{g/L}$. They found mortality increased as concentration increased from 20-80 $\mu\text{g/L}$; but there was no further increase in effects at concentrations above 80 $\mu\text{g/L}$. Their threshold is approximately twice the mean maximum concentration expected in lotic water after operational aerial spraying.

Only five experimental lentic studies of fenitrothion have been reported (Tables 4.12 and 4.13). Three dealt with direct application of fenitrothion to lakes during forestry operations (Kingsbury 1977a, 1978a, 1978b), one was an operational agricultural application (a low-level aerial spray) (Thirumurthi et al. 1975), and one was an experimental ground application (Fairchild, in prep). The limited effects on lakes, documented by the studies of direct

Table 4.9 Effects of fenitrothion on aquatic invertebrate drift in field experiments, based on dosage rate and insecticide residue analysis.

<i>Residue level (Dosage rate g a.i./ha)</i>	<i>Measured effect</i>	<i>Reference</i>
(2 x 280)	increase Simuliidae drift in one stream; most streams short-lived increase in drift, but numbers influenced by spates	Kingsbury 1978a
1,000 µg/L	large increase in drift; different pat- terns among species studied	Hasegawa et al. 1982
4.06 µg/L	increased drift, Plecoptera, Chironomidae, and Simuliidae 2-3 h after spray	Holmes 1982a
15 µg/L	increased drift; drift of affected organisms greater near the bottom than at stream surface	Eidt and Weaver 1984a
21.45 µg/L	increased drift, Plecoptera, Ephemeroptera, Trichoptera, and Chironomidae after 2 h	Holmes 1982a
10-23 µg/L for 3 h	large increase in drift; most kill consisted of <i>Leuctra</i> spp.	Eidt 1981
26.1 µg/L about 20 µg/L for 4 h	several-fold increase in drift, mostly Plecoptera and Ephemeroptera organisms buried in substrate had less mortality	Eidt 1978
31 µg/L (280)	rapidly increased drift, peak about 3 h; returned to pre-spray levels within 24 h	Poirier and Surgeoner 1986
54-159 µg/L	lethal threshold to benthic community greater than or equal to 20 µg/L; no increase in mortality above 80 µg/L increased drift and reduced benthos	Ahern and Leclerc 1981
100-460 µg/L for 1.3 h	<i>Baetis</i> and <i>Nemoura</i> drifted early; <i>Elmis</i> and <i>Ephemera</i> drifted late	Yasuno et al. 1981
90-500 µg/L for 24 h	increased drift; Ephemeroptera and Plecoptera reacted quickly; most groups peaked in first 12 h then declined	Peterson and Zitko 1974

Table 4.10 Effects of fenitrothion on benthic invertebrates in lotic ecosystems after aerial spraying, based on dosage rate (with no corollary insecticide residue analysis).

<i>Dosage rate g a.i./ha</i>	<i>Measured effect</i>	<i>Reference</i>
1 x 140	no evidence of severe post-spray mortalities using substrate and Surber samples	Elson et al. 1972
1 x 140	benthic populations increased; Surber samples; small sample size	Kingsbury and Sarrazin 1975
1 x 140	possible short-term (<12 d) reduction in benthos using Surber samples	Kingsbury and Sarrazin 1975
2 x 140	no reductions attributable to spraying	Penney 1971b
2 x 140	no significant reductions in benthic populations using Surber samples	MacDonald and Penney 1969
2 x 140	no significant reductions in benthic populations using Surber samples	MacDonald and Penney 1969
2 x 140	no significant reductions in benthic populations using Surber sample	Buckner and McLeod 1977
2 x 140	no decline in benthos	Dimond and Malcolm 1971
2 x 140	one brook out of six studied had effects; benthic chironomids significantly reduced; other groups non-significant reductions; total unchanged	Dimond and Malcolm 1971
2 x 140	numbers in Surber samples fell 30-50%; benthos numbers on artificial substrates fell 70% for Ephemeroptera, Plecoptera, Trichoptera	Elson et al. 1972
3 x 140	no significant reductions in benthic populations using Surber samples	MacDonald and Penney 1969
1 x 210	no change in benthic diversity; decrease in Simuliidae and Ephemeroptera, not significant cant; no other apparent differences; recovered within 30 d	Rabeni and Gibbs 1976b

(continued)

Table 4.10 (continued)

<i>Dosage rate g a.i./ha</i>	<i>Measured effect</i>	<i>Reference</i>
1 x 210	Plecoptera reduced, total biomass unchanged	Penney 1971a
2 x 210	no effects attributable to spraying	Riche and Coady 1969
2 x 210	no significant reductions in benthos; possible decrease, number too small, variance high	Buckner and McLeod 1977
2 x 210	no effects related to spray	Montreal Engineering Co. 1976
2 x 210	no clear result for benthos	Riche and Coady 1969
2 x 210	possible effect in benthos, none at 30 d	Montreal Engineering Co. 1978
2 x 210	appears first spray substantially decreased benthic populations; several groups affected; low reliability sampling, Surber samples	Buckner et al. 1976
1 x 280	biomass decrease >1/3	Welch in Symons 1977a
1 x 420	biomass decrease; large loss of stoneflies; no reduction in Diptera, one stream studied	Vincent 1969 in Dimond and Malcolm 1971

Table 4.11 Effects of fenitrothion on benthic invertebrates in lotic ecosystems after aerial spraying, based on residue analysis.

<i>Peak concentration (Dosage rate g a.i./ha)</i>	<i>Measured effect</i>	<i>Reference</i>
0.25 µg/L	no effects on benthos; possible increase in Simuliidae and Chironomidae after second spray; no control	Holmes 1982a
1.33 µg/L (210)	no depletion of benthos	Eidt 1975a
1.74 µg/L (210)	no significant effect on benthos	Holmes 1979
1.84 µg/L (2 x 210)	Ephemeroptera reduced 2 d after spray other groups appeared to decline; not significant; total benthic standing crop unchanged	Holmes 1982a
2.0 µg/L (2 x 210)	no effect on benthos	Eidt 1977
3.13 µg/L (2 x 210)	no effect on benthos	Eidt 1977
5.25 µg/L (210)	no effect on benthos	Eidt 1975a
5.25 µg/L (2 x 210)	no effect on benthos	Eidt 1977
6.38 µg/L (210)	no effect on benthos	Eidt 1975a
3.0-6.7 µg/L (2 x 140)	no effect on benthos	Tracy and Purvis 1976
11.2 µg/L (2 x 140)	20 stone count; no benthos reduction; Surber showed slight decline in benthos with rapid recovery for most groups	Hall and Eaton 1973
7-14 µg/L (2 x 140)	Surber samples showed no decline in benthos (late summer adulticide application)	Banks and Hall 1972

(continued)

Table 4.11 (continued)

<i>Peak concentration (Dosage rate g a.i./ha)</i>	<i>Measured effect</i>	<i>Reference</i>
7.4-14.3 µg/L (2 x 140)	20 stone count indicated decline in benthos; (late summer adulticide application)	Banks and Hall 1972
31.0 µg/L (210)	no effect on benthos	Tracy and Purvis 1976
38 µg/L (2 x 210)	no effect on benthos	Lord et al. 1978
42 µg/L (2 x 280)	no effect on benthos	Lord et al. 1978
4.2-48 µg/L (300)	qualitatively many taxa survived treatment	Wells et al. 1978a
39-49 µg/L (2 x 210)	70-80% reduction in benthos; Ephemeroptera eliminated; few signs of recovery one month after spray	Coady 1978
64 µg/L (280)	no decrease in standing crop recorded	Flannagan 1973

Table 4.12 Effects of fenitrothion on benthic invertebrates in field experiments based on dosage rate (with no corollary insecticide residue analysis).

<i>Dosage rate g a.i./ha</i>	<i>Measured effect</i>	<i>Reference</i>
Lotic		
2 x 280	no depletion of benthos related to treatments	Kingsbury 1978a
1 x 560	reduced numbers and weight of benthos by 2/3 based on Surber samples (n=20)	Penney and MacDonald 1966
1,000 µg/L	many benthic taxa before spray, only two after	Hasegawa et al. 1982
Lentic		
2 x 280	no dramatic effects on lake zooplankton, or benthos; significant decrease in shallow dwelling; Ephemeroptera; no control; no evidence of mortality with visual check	Kingsbury 1978a
500 low level aerial appl. to rice paddy	100% mortality of frogs, mosquitos, dragonflies, and water boatmen; a few surviving crabs and beetles (operational spray)	Thirumurthi et al. 1975
Other		
0.31-9.6 µg/L	mosquito and midge inhabitants of pitcher plant leaves were found to be susceptible	Fairchild et al. 1987

Table 4.13 Effects of fenitrothion on benthic invertebrates in field experiments, based on insecticide residue analysis.

Residue concentration	Measured effect	Reference
Lotic		
4.06 µg/L	Ephemeroptera and Trichoptera reduced in benthos; total standing crop unchanged	Holmes 1982a
21.45 µg/L	no reduction any taxa; no reduction in benthic standing crop	Holmes 1982a
10-23 µg/L for 3 h	benthic standing crop reduced about 50% most kill consisted of <i>Leuctra</i> spp.; recovery complete within 50 d	Eidt 1981
26.1 µg/L about 20 µg/L for 4 h	reduction of benthos; organisms buried in substrate had less mortality	Eidt 1978
54-159 µg/L	lethal threshold to benthic community greater than or equal to 20 µg/L; no increase in mortality above 80 µg/L; increased drift and reduced benthos	Ahern and Leclerc 1981
100-460 µg/L for 1.3 h	benthos reduced 80-90% for two weeks recovery monitored; <i>Simulium</i> increased to above pretreatment; <i>Anisogammarus</i> eliminated	Yasuno et al. 1981
Lentic		
3.8-5.8 µg/L	little impact on lake biota; surface populations of zooplankton reduced for a short time; <i>Chaoborus</i> disappeared; no significant change in benthos, Chironomidae and Amphipoda did decline, not significant	Kingsbury 1978b
21.6 µg/L 2.14 µg/L after 24 h	possible slight decrease in benthos; qualitative observations on mortality of Odonata nymphs in shallow water (<1 m); possible decrease in elmids populations	Kingsbury 1977b
42-81 µg/L	benthic biomass reduced about 50% chironomids and ceratopogonids main groups, others affected include Coleoptera, Hemiptera, and Odonata, Gyrinidae eliminated; over 12 months required for recovery	Fairchild in prep

applications at rates used in forestry, indicate that operational sprays should have only very minor effects on large lakes. In operational forest-spray programs, lakes are protected by no-spray buffer zones which limit the amount of insecticide reaching the water. Agricultural application to shallow water (at double forestry rates), caused near 100% mortality of the benthos (Thirumurthi et al. 1975). The last study highlights areas potentially sensitive to forest spraying. At residue concentrations (42-81 $\mu\text{g/L}$) which were comparable to those recorded after operational aerial application of 210 g a.i./ha (Tables 4.3 and 4.5), about a 50% reduction in numbers of benthic invertebrates was recorded on small bog ponds experimentally sprayed with a mist blower (Fairchild in prep). That such residues are operationally realistic is further evidenced in that they are identical to the calculated concentration in a 10-cm. water column from a direct 210 g a.i./ha application, assuming 40% of the emitted pesticide is deposited. Recovery of benthos populations to control densities required greater than one year. The shift in benthic community structure which accompanied the reduction in standing crop, decreased the energy and nutrient flux caused by insects emerging from the ponds. The amount of energy and nutrient cycling within the ponds increased while the contribution to the surrounding terrestrial system was reduced. It was also demonstrated (Fairchild et al. 1987) that mosquito and midge larvae inhabiting pitcher plant leaves are sensitive to levels of fenitrothion which might be encountered in field applications.

4.9 PROBABILITY OF MEASURABLE DISTURBANCE

Most studies indicate limited perturbations due to fenitrothion aerial sprays. However, ecologically significant effects occasionally do occur.

There now is a mass of literature from which a set of heuristics can be derived making it possible to gauge the risk to a particular water body from aerially-applied fenitrothion. A simple literature-based approach would at least give some idea of proportions or probabilities of effects after operational spraying. That has been attempted below. The results are based on the entries in Tables 4.7 and 4.8 for drift, and Tables 4.10 and 4.11 for benthos.

In 24 of 34 drift-monitoring studies (Tables 4.7 and 4.8), an increase in drift after spraying was reported.

If the sites represent a cross section of streams in forest habitat, it could be inferred that 70% of streams experience some increased drift after aerial spraying of fenitrothion. That conclusion must be tempered by the realization that most of the studies were done between 1969 and 1980, during which time buffer-zone strategies were evolving and not widely implemented (Kingsbury and Trial 1987). The effect that the implementation of buffer zones may have on mitigating aquatic impacts is discussed in Section 4.1.4. Only one drift-monitoring study has been done since 1980 (Holmes 1982a). It showed very limited effects in one stream (increased duration of drift, not magnitude) and none in another.

Monitoring of benthic invertebrate populations (Tables 4.10 and 4.11) indicated that six of 41 entries recorded decreases in benthic populations after spraying. Again, if the sites are representative, it could be inferred that at least 15% of streams will experience some measurable reduction in benthos after aerial spraying of fenitrothion. It must again be noted that all of those studies, except that by Holmes (1982a), were done prior to 1980 and do not reflect the consequences of implementation of current buffer zone tactics in most jurisdictions.

As previously discussed, the diversity of response to fenitrothion spraying by invertebrate communities is caused by many factors which, acting together, determine a particular aquatic ecosystem's chances of sustaining significant ecological effects. Protection by the forest canopy is an important factor affecting spray reaching water bodies. Some studies, which demonstrated severe effects from spraying, indicated a lack of forest cover (Coady 1978); however, most authors have not included information on surrounding vegetation.

The presence of pools along streams may also contribute to increased ecological effects. The pools collect more fenitrothion deposit, and release it into the stream slowly, increasing exposure. If the pool is large enough, it may have less forest-canopy cover than the narrower sections of streams.

Characteristics of the water are also important. Streams that drain from boggy areas with dark, peaty water and low pH may have reduced rates of photolytic and hydrolytic breakdown, and so retain fenitrothion longer (NRCC 1975). Those characteristics are probably important in contributing to the ecological effects observed in bog ponds (Fairchild, in prep).

All of those factors are probably insignificant compared to the creation and proper implementation of buffer zones around sensitive habitats which limit the amount of fenitrothion reaching off-target areas.

4.10 LABORATORY STUDIES WITH FISH

4.10.1 Acute Toxicity

The acute toxic effects of fenitrothion have been determined for a wide variety of fish species. The LC50s (24-96 h) for warm-water fish (Table 4.14) range from 0.7-10.7 mg/L, with bluegill sunfish (*Lepomis macrochirus*) and medaka (*Oryzias latipes*) being the most sensitive species tested. According to the data of Sanders et al. (1983), the liquid formulation (presumably the emulsifiable concentrate) is generally more toxic, on an active-ingredient basis, than the technical material which, in turn, is more toxic than the wettable powder.

The toxicity of fenitrothion to cold-water fish (Table 4.15) is very similar (0.4-11.0 mg/L) to its toxicity to warm-water species. It appears that the decreasing order of toxicity of the "out of the container" formulations is: emulsifiable concentrate, technical material, and wettable powder (Sanders et al. 1983). In addition, testing of the conventional spray mixtures used in forestry programs (Wells et al. 1978b) indicated that the water emulsion (fenitrothion/Arotex/Atlox/water) is four to five times more toxic to fish than the oil solution (fenitrothion/Arotex/585 oil). Although the above investigators also observed that one of the components of the spray solution, Arotex 3470D, was about half as toxic as the active ingredient itself, they judged that the primary toxicity of spray solutions containing Arotex 3470D was almost entirely due to the fenitrothion content. The newer Sumithion 180F formulation, consisting of finely milled fenitrothion particles in suspension, was toxic to rainbow trout (*Salmo gairdneri*) at almost the same concentrations, on an active-ingredient basis, as technical fenitrothion (W. Ernst et al., in prep.).

Klaverkamp et al. (1977) and Takimoto et al. (1984a) observed that for rainbow trout and the medaka, sensitivity to fenitrothion increased during the developmental stage from embryo to fingerling or adult (Table 4.16). Life-stage tests conducted by Wildish et al. (1971) and Hashimoto et al. (1982), on the other hand, indicated little susceptibility differences in various life stages of carp (*Cyprinus carpio*) or Atlantic salmon (*Salmo salar*). Within the

same age class, the smaller size ranges of rohu (*Labeo rohita*) are significantly and progressively more sensitive to fenitrothion than are larger fish (Murty et al. 1983).

Table 4.17 lists the relative toxicity of fenitrothion and its various theoretical degradation products to medaka, as measured by Miyamoto et al. (1978). With the exception of 3-methyl-4-aminophenol, which is not a major degradation product, all compounds were less toxic than the parent fenitrothion. Zitko and Cunningham (1975) also determined that the S-methyl isomer of fenitrothion is as acutely toxic to Atlantic salmon as is fenitrothion, even though it has a greater acetylcholinesterase (AChE) depressing affect.

4.10.2 Bioaccumulation

Fenitrothion has a relatively high water solubility (38.7 mg/L at 20-25° C) and a moderate octanol/water partition coefficient (2767 to 3380) (Chiou et al. 1977; Kanazawa 1981). Since the tendency of organisms to accumulate xenobiotics is dependent on those factors, as well as the lipid content in fish (Takimoto et al. 1984b), it is not surprising that fish accumulate fenitrothion moderately. During acute laboratory exposures, the bioconcentration factors (BCF is the concentration in wet weight of tissue divided by the exposure concentration) for various fish species ranged from 30 to 6058 (Table 4.18). The rate of uptake of fenitrothion is apparently metabolically related because it is more rapid at higher temperatures. However, the maximum BCFs do not change substantially with temperature (Takimoto et al. 1987). Bioconcentration factors for medaka and mullet (*Mugil cephalus*) were also significantly higher in a salinity of 23 ‰ than in freshwater exposures (Takimoto et al. 1987). Lockhart et al. (1982) determined that the bioconcentration characteristics of fenitrothion in rainbow trout larvae closely approximate the generalized equation:

$$\log \text{bioconcentration} = 0.454 (\log \text{weight of fish}) - 0.049 (\log \text{time}) + 0.049 (\text{water characteristic factor}) + 0.007.$$

Under flow-through conditions, fenitrothion in medaka exposed to 0.1 mg/L reached concentrations as high as 39 mg/L (wet weight) (Takimoto et al. 1984b). While uptake of fenitrothion by fish rapidly reaches a maximum within 24 h (Lockhart et al. 1982) to 3 d (Kanazawa 1975; Takimoto et al. 1984b), its clearance is equally rapid, with half-lives reported to be between 3 h (Takimoto et al. 1987) and 2 d (Lockhart et al. 1982; Takimoto et al. 1984b).

Table 4.13 Acute toxicity of fenitrothion to various warmwater fish species exposed under static conditions.

Species	Formulation	Toxic concentrations			Reference
		24 h LC50 (mg/L)	48 h LC50 (mg/L)	96 h LC50 (mg/L)	
Medaka (<i>Oryzias latipes</i>)	Technical			3.5	Takimoto et al. 1987
Dwarf catfish (<i>Mystus cavasius</i>)	Technical			3.3	Murty et al. 1983
Rohu (<i>Labeo rohita</i>)	Technical			2.8	Murty et al. 1983
Carp (<i>Cyprinus carpio</i>)	—	4.0			Hashimoto et al. 1982
Labyrinth catfish (<i>Clarius batrachus</i>)	EC 50			10.5-10.7	Sharma et al. 1981
Tilapia (<i>Tilapia mossambica</i>)	EC 50		6.0		Koundinya et al. 1978
Rohu	EC 50	6.0	5.8	5.4	Sharma et al. 1981
Medaka	Technical			1.0	Takimoto et al. 1984a
Channel catfish (<i>Ictalurus punctatus</i>)	Technical Liquid Wettable powder			4.3 8.6 5.1	Sanders et al. 1983
Bluegill sunfish (<i>Lepomis macrochirus</i>)	Technical Liquid Wettable powder			1.0 0.7 1.4	Sanders et al. 1983
Fathead minnows (<i>Pimephales promelas</i>)	Technical Liquid Wettable powder			3.2 2.4 4.2	Sanders et al. 1983
Fathead minnow Embryos and Larvae	—	3.3-8.2			Kleiner et al. 1984

Table 4.15 Acute toxicity of fenitrothion formulations and components of formulations to various coldwater fish species.

Species	Formulation	Toxic concentrations			Conditions	Reference
		24 h LC50 (mg/L)	48 h LC50 (mg/L)	96 h LC50 (mg/L)		
Rainbow trout (<i>Salmo gairdneri</i>)	Fenit/Arotex Atlox/water			2.4 (2.0-2.8)*	Static/ fingerling	Wells et al. 1978b
Rainbow trout	Fenit/Arotex/ 585 oil			11 (5.6-28)*	Static/ fingerling	Wells et al. 1978b
Rainbow trout	Arotex 3470 D			3.8 (2.7-5.0)	Static/ fingerling	Wells et al. 1978b
Rainbow trout	Atlox 3409 F			24-56	Static/ fingerling	Wells et al. 1978b
Rainbow trout	Technical			1.9 (1.0-3.2)	Static/ fingerling	Wells et al. 1978b
Rainbow trout	Technical	3.4	2.0		Flow-through/ fingerling	Klaverkamp et al. 1977
Rainbow trout		2.7			Flow-through/ fingerling	Duangasawadi and Klaverkamp 1979
Rainbow trout	Technical			1.8 (1.1-2.5)	Static/ fingerling	Doe et al. 1988
Cutthroat trout (<i>S. clarki</i>)				2.9	Static/ fingerling	Woodward and Mauck 1980
Rainbow trout	Fenit/Triton X-100/water	3.6-3.7		2.4-2.5**	Static/ fingerling	Holmes 1982b
Rainbow trout	Fenit/Cyclosol 63-Triton X-100/ water	3.6		2.4-2.5	Static/ fingerling	Holmes 1982b
Rainbow trout	Technical Liquid Wettable powder			1.0 0.7 1.4	Static	Sanders et al. 1983

(continued)

Table 4.15 (continued)

Species	Formulation	Toxic concentrations			Conditions	Reference
		24 h LC50 (mg/L)	48 h LC50 (mg/L)	96 h LC50 (mg/L)		
Rainbow trout	—		1.3-1.8		Recirculating/ fingerling	Thellan et al. 1982
Brook trout (<i>Salvelinus fontinalis</i>)	—		1.4-1.8		Recirculating/ fingerling	Thellan et al. 1982
Atlantic salmon (<i>Salmo salar</i>)	—		0.9-1.5		Recirculating/ parr	Thellan et al. 1982
Atlantic salmon	Technical	4.2- 4.7	2.7	0.9	Static/fry	Wildish et al. 1971
Rainbow trout	Technical			1.7	Static/ fingerling	W. Ernst (in prep)
	Flowable**			2.2		
Threespine stickleback (<i>Gasterosteus aculeatus</i>)	Technical			3.1	Static/adult	W. Ernst (in prep)
	Flowable**			3.1		

* LC50 stated as mg of formulation per litre, in all other cases LC50 stated as mg active ingredient per litre.

** Fenitrothion flowable consists of finely milled fenitrothion particles in suspension.

Table 4.16 Acute toxicity of fenitrothion to various life stages of fish.

Species	Life stage	24 h LC50 (mg/L)	96 h LC50 (mg/L)	Test conditions	Reference
Rainbow trout (<i>S. gairdneri</i>)	Embryo	> 34.0		flow-through	Klaverkamp et al. 1977
	Sac Fry				
	0-5 days	> 34.0			
	6-9 days	34.0			
	10-11 days	< 34.0			
	Fingerlings	3.4			
Medaka (<i>Oryzias latipes</i>)	Embryo		> 10	static	Takimoto et al. 1984a
	Post-larva		2.36		
	Adult		1.0		
Carp (<i>Cyprinus carpio</i>)	Eyed egg	3.5		static	Hashimoto et al. 1982
	Sac fry	1.5			
	Floating fry	1.7			
	Feeding fry				
	-one week	2.2			
	-two weeks	2.8			
	-four weeks	3.8			
	-eight weeks	2.6			
	-eleven weeks	4.0			

Table 4.17 Toxicity of various fenitrothion degradation products to fish.

Species	Compound	LC50
Medaka (<i>Oryzias latipes</i>)	Fenitrothion	3.4
	3-Hydroxymethyl-	9.2
	3-Formyl-	8.6
	3-Carboxy-	> 10.0
	Fenitrooxon	> 10.0
	3-Hydroxymethyl-	> 10.0
	3-Formyl-	> 10.0
	3-Carboxy-	> 10.0
	S-Methylfenitrothion	> 10.0
	Aminofenitrothion	> 10.0
	N-Formyl-	> 10.0
	N-Acetyl-	> 10.0
	Aminofenitrooxon	> 10.0
	N-Formyl-	> 10.0
	N-Acetyl-	> 10.0
	Desmethylfenitrothion	> 10.0
	Desmethyl-f-nitrophenol	8.4
	3-Methyl-f-nitrophenol	> 10.0
	3-Hydroxymethyl-f	> 10.0
	3-Formyl-4-	> 10.0
	3-Carboxy-4-	> 10.0
	3-Methyl-4-Aminophenol	0.13
	3-Formyl-4-	> 10.0

Tests conducted by Miyamoto et al. 1978.

4.10.3 Water Quality and Toxicity

The acute lethal effects of fenitrothion are not significantly altered by changes in water hardness ranging from 40-320 mg/L of CaCO_3 (Woodward and Mauck 1980). Unlike pesticides which are either acidic or basic in solution, fenitrothion does not change in toxicity as the pH of water changes because it does not ionize in solution (Woodward and Mauck 1980; Doe et al. 1988).

Woodward and Mauck (1980) were unable to distinguish changes in fenitrothion toxicity to cutthroat trout (*Salmo clarki*) over temperatures ranging from 7-12° C. Other investigators (Duangsawasdi and Klaverkamp 1979) observed a substantial decrease in the rainbow trout 24 h LC50 (from 4.3-1.9 mg/L) over temperatures ranging from 8-22° C.

4.10.4 Sublethal Effects

In addition to lethality studies, there have been a number of laboratory investigations attempting to determine the physiological response of fish to fenitrothion exposure.

4.10.4.1 Metabolism

Fish are able to metabolize fenitrothion. They do that primarily by methylation of the P-O-alkyl bonds, hydrolysis of P-O-aryl linkages, and oxidation of P-S bonds (Miyamoto et al. 1978; Takimoto et al. 1987). The major metabolites which have been proposed or measured include: 3-methyl-4-nitrophenol and its glucuronide oxide (Takimoto and Miyamoto 1976; Anjum and Quadri 1986; Takimoto et al. 1987); fenitrooxon (Klaverkamp and Hobden 1980; Anjum and Quadri 1986; Takimoto et al. 1987); and desmethyl fenitrooxon (Takimoto and Miyamoto 1976; Takimoto et al. 1984b; Anjum and Quadri 1986). Those metabolites are less acutely toxic to medaka than the parent compound (Miyamoto et al. 1978).

Radiography of medaka exposed to ^{14}C labeled fenitrothion has demonstrated that prior to hatch, fenitrothion is concentrated in the yolk sac. In post-larval to adult stages, it is concentrated in the internal organs, primarily the liver (Takimoto et al. 1984b, 1987). Both the high concentrations of the parent pesticide in the liver and the presence of metabolites in that organ (Anjum and Quadri 1986) are good indications that the liver is the primary site for biotransformation of fenitrothion (Klaverkamp and Hobden 1980).

4.10.4.2 Acetylcholinesterase inhibition

Organophosphorus pesticides exert their primary effect by selectively binding to active sites of the AChE enzyme which subsequently interferes with the synaptic transmission of nerve impulses. That action is probably the primary mechanism for toxicity in fish (Zitko et al. 1970). No direct relationship is evident, however, between AChE depression and toxic effects. The S-methyl isomer of fenitrothion (a low level contaminant in commercial products) is toxic to fish at approximately the same concentrations as fenitrothion (Zitko and Cunningham 1975) even though it is a much more potent inhibitor of AChE by 100 to 1 000 times (Kovacicova et al. 1973). One explanation is that S-methyl fenitrothion has a much more rapid hydrolysis rate and thus disappears more quickly from exposure solutions (Kovacicova et al. 1973). However, some fenitrothion metabolites, such as fenitrooxon, are also more potent AChE inhibitors (Klaverkamp and Hobden 1980) but are less acutely toxic in themselves (Miyamoto et al. 1978). Direct correlations between fenitrothion exposure concentrations and brain AChE depression have been difficult to obtain (Witty 1985).

The *in vitro* rainbow trout studies of Klaverkamp and Hobden (1980) indicated that fenitrothion was not activated to a more potent AChE inhibitor by incubation with rainbow trout liver homogenate. Increased toxicity of fenitrothion by elevated temperatures (Duangsawasdi and Klaverkamp 1979) was due not only to increased metabolism of fenitrothion but also to increased rates of uptake as observed by Takimoto et al. (1987). Even though heart and serum may be more sensitive, the brain has been used most often as the tissue for measuring AChE activity because of smaller variability of normal activity levels in this tissue (Duangsawasdi and Klaverkamp 1979; Lockhart et al. 1973).

4.10.4.3 Biochemical/histopathological changes

Several attempts have been made to measure sublethal stress effects on fish through changes in their biochemistry. When exposed to fenitrothion at 6 mg/L for 48 h, *Tilapia* (*Tilapia mossambica*) had elevated succinate dehydrogenase activity and decreased lactate dehydrogenase activity, thus indicating an increased reliance on anaerobic metabolism (Koundinya and Ramamurthi 1978). Investigators have documented conditions indicating stress-induced hyperglycemia in fenitrothion-exposed fish. One example is the increased serum glucose and phosphorylase activity and decreased hepatic glycogen in *Tilapia* exposed to 6 mg/L

Table 4.18 Bioconcentration factors (BCF)* in fish exposed to fenitrothion under laboratory conditions.

Species	Time duration	Exposure conc. (mg/L)	Conditions	Maximum BCF	Reference
Medaka (<i>Oryzias latipes</i>)	7 d	0.1	Flow-through 25°	255	Takimoto et al. 1976
	7 d	0.1	Freshwater Flow-through 15°	339	
	7 d	0.1	Freshwater Flow-through 25° Seawater	303	
Mullet (<i>Mugil cephalus</i>)	7 d	0.1	Flow-through 25°	30	
	7 d	0.1	Freshwater Flow-through 25° Seawater	179	
Topmouth gudgeon (<i>Pseudorasbora parva</i>)	14 d	.005-.020	Flow-through	6058	Kanazawa 1981
Topmouth gudgeon	28 d	0.81	Static	203	Kanazawa 1981
Rainbow trout (<i>Salmo gairdneri</i>) Larvae	24 h	0.005	Static	116	
Medaka "Post larva"	3-7 d	0.1	Flow through	540	Takimoto et al. 1984b

* BCF is the concentration in fish (wet wt.) divided by exposure concentration.

fenitrothion for 48 h (Koundinya and Ramamurthi 1979b). Another is the elevated alkaline phosphatase activity in gill, kidney, liver, muscle, intestine, and brain of *Sarotherodon mossambicus* exposed to the same concentration (Koundinya and Ramamurthi 1982).

Saxena and Mani (1985) observed that freshwater murrel (*Channa punctatus*) exposed to 0.75 mg/L fenitrothion had significantly decreased protein-bound iodine in blood after 60 d, suggesting a possible thyroid hormone mediated growth effect. During 30-d exposures at concentrations of 0.13 mg/L, Kleiner et al. (1984) found no weight change in fathead minnow (*Pimephales promelas*) embryos or larvae.

There is some indication that fenitrothion causes metabolic depression in fish. Koundinya and Ramamurthi (1979a) determined that tissues from Tilapia, which survived exposure to 6 mg/L for 48 h (LC50 concentration), had markedly reduced oxygen consumption. Oxygen consumption was reduced by as much as 56% compared with controls and the tissues affected, in decreasing order, were: gill, liver, brain, muscle, intestine, and kidney. Murty et al. (1983) measured a significant, and dose dependent, reduced-oxygen uptake by rohu exposed to fenitrothion for 2 h. The threshold for producing that effect was 2.0 mg/L.

In rainbow trout exposed to fenitrothion, several investigators noted responses such as decreased heart rate, increased respiration rate, and increased buccal amplitude, all of which may be related to decreased tissue oxygen consumption (Klaverkamp et al. 1977; Duangawasdi and Klaverkamp 1979). The former authors noted those effects at exposures to 2.0 mg/L while the latter observed them at exposures of 0.05 mg/L over 24 h.

Little effort has been devoted to determining if there are any histopathological changes in fenitrothion-exposed fish. In a very limited study (single individuals sampled at each sampling time), it was observed that during static exposures for 45-90 d at 1.0 mg/L fenitrothion, the internal organs of labyrinthic catfish (*Clarius batrachus*) suffered degradative changes at the cellular level (Mandal and Kulshrestha 1980). Liver cells became progressively more vacuolated and many had multinucleate cells. In the kidney, glomeruli underwent shrinkage and the epithelial cells of uriniferous tubules became desquamated and vacuolated, with the eventual rupture of tubules. In the intestine, the epithelial cells of villi became vacuolated and villi eventually ruptured.

4.10.4.4 Reproductive effects

The reproductive effects of fenitrothion on fish have been the subject of a number of investigations (Table 4.19). The effects noted are: a decrease in the gonadosomatic index (Saxena and Garg 1978; Kapur and Virk 1983; Pawar and Katdare 1983); an inhibited gonadal development characterized by a decrease in the number of mature oocytes (Kapur and Virk 1983; Pawar and Katdare 1983; Saxena and Garg 1978; Saxena et al. 1986); an inhibited production of steroid (steroidal dehydrogenase 3 PHSD) (Kapur et al. 1978); and a decrease in the number of live young produced in the viviparous guppy (*Poecilia reticulata*) (Yasuno et al. 1980). Saxena et al. (1986) reported that retardation of gonadal development occurred within 30 d in the freshwater murrel exposed to 1.5 mg/L fenitrothion. That was measured by a decrease in gonadal protein, RNA, ascorbic acid, and total lipid content as well as an increase in phospholipid and cholesterol content. Only exposure concentrations of 0.25 mg/L fenitrothion or greater and test durations of 30 d or longer produced those effects.

Fenitrothion exposure affects the embryo development of fish (Table 4.20). Hiraoka and Okuda (1984a) observed a significantly higher incidence of vertebral centrum damage in medaka eggs exposed to 0.4 mg/L fenitrothion for one week than in controls. Carp eggs incubated in fenitrothion solutions for 55-74 h had a dose-related and significantly greater proportion of deformities, primarily related to spinal development. That response had a threshold of approximately 0.25 mg/L (Kamaldeep and Tbor 1977). Those investigators also determined that under the same exposure conditions, the LC50 for hatching was approximately 0.65 mg/L.

In separate studies, the viviparous female guppy, when exposed to fenitrothion at concentrations of 0.25 mg/L or greater for two mo (Yasuno et al. 1980) and 10 mg/L for 4 h (Miyashita 1984), produced significantly smaller numbers of live young. In addition, Yasuno et al. (1980) determined that while the threshold for measuring significantly higher rates of abortion was approximately 0.25 mg/L fenitrothion, the rate of abortion was 100% at 1.0 mg/L. Casual observations (Takimoto et al. 1984b) indicate that fenitrothion exposure may result in damage to circulatory organs of medaka embryos. However incidence, threshold, and dose response for such effects are not well defined. Wildish et al. (1971) observed some unusual swelling of the head as well as spinal deformities in Atlantic salmon fry exposed to fenitrothion. They indicated that the possibility of

Table 4.19 Reproductive effects observed in fish exposed to fenitrothion under static conditions.

Species	Measured Concentration (mg/L)	Duration	Observation	Reference
Murrel (<i>Channa punctatus</i>)	1.5	150 d	- Decrease in ovarian weight - Increase in number of undeveloped oocytes	Saxena and Garg 1978
Carp (<i>Cyprinus carpio</i>)	0.5	6 mo	- Inhibition of steroidogenesis (steroidal-dehydrogenase 3- HSD) in gonadal tissue	Kapur et al. 1978
Sucking barbs (<i>Garra mullya</i>)	1.0	30 d	- Ovarian weight one-quarter of controls- No mature oocytes - Reduced number of immature oocytes	Pawar and Katdare 1983
Carp	1.5	120 d	- Significant decrease in ovarian weight - Decrease in mature oocytes - Increase in immature oocytes	Kapur and Virk 1983
Murrel	1.5	120 d	- By 30 d - decrease in ovary and testes; development	Saxena et al. 1986
Guppy (<i>Poecilia reticulata</i>)	0.25	2 mo	- Decrease in number of young	Yasuno et al. 1980

Table 4.20 Effects on development of embryos when exposed to fenitrothion.

Species	Concentration (mg/L)	Duration	Condition	Observations	Reference
Medaka (<i>Oryzias latipes</i>)	0.4	1 w	Static	- Approximately 3 times vertebral centrum damage of controls.	Hiraoka and Okuda 1984a
Guppy (<i>Poecilia reticulata</i>)	10.0	4 h	Static	- Twice dead birth and abnormal embryos of controls.	Miyashita 1984
Guppy	0.25 1.01	2 mo	Semi-static	- Decrease in live young, higher rate of abortion - All females aborted	Yasuno et al. 1980
Medaka	2.0	8 d	Static	- Damaged circulatory organs. - Reduced hatchability	Takimoto et al. 1984a
Catfish (<i>Heteropneustes fossilis</i>)	25.0	33 h	Semi-static	- Greater number of abnormal embryos.	Sur et al. 1979
Carp (<i>Cyprinus carpio</i>)	0.65 0.25	55-74 h 55-74 h	Static Static	- LC50 for live hatch. - Threshold for producing deformed larvae.	Kamaldeep and Toor 1977

the acetone carrier being the cause of the malformation could not be discounted. Some of those effects may relate to the observations of Miyamoto et al. (1978) that the 3-hydroxy-methyl fenitrothion metabolite was positively mutagenic in a test on *Salmonella typhimurium*.

4.10.4.5 Behavioural effects

A number of studies have attempted to measure the behavioural effects of fenitrothion on fish.

After statically exposing brook trout to various fenitrothion concentrations for 24 h, Peterson (1974) observed that when the fish were placed in flowing-water exercise chambers, maximum swimming velocities were reduced in a dose-dependent manner. The lowest concentration at which a significant reduction in swimming velocity was observed was 0.5 mg/L, a concentration near the lethal threshold for that species. A concentration of 0.1 mg/L for 48 h was sufficient to produce a similar effect (Thellan et al. 1982) in the same species.

Scherer (1975), using a counter-current chamber, demonstrated that goldfish (*Carassius auratus*) were able to avoid water contaminated with fenitrothion. The threshold for avoidance was 0.01 mg/L. The ecological implication of that observation is that fish may be able to sense and avoid lethal concentrations of the insecticide. In sprayed streams, such a response has the potential to alter the spatial structure of fish populations.

Several early studies attempted to measure fenitrothion effects on learning ability and predation. Hatfield and Johansen (1972) observed that the learning ability of Atlantic salmon parr, measured by response to shuttlebox conditioning using light and electric shock, was totally inhibited by 24-h exposures to 1.0 mg/L but entirely unaffected by exposures to 0.1 mg/L. Hatfield and Anderson (1972) observed that Atlantic salmon fry exposed to fenitrothion at 1.0 mg/L suffered significantly greater predation by brooktrout in test tanks than unexposed fish, but that there was no effect at 0.1 mg/L. Atlantic salmon fry exposed to fenitrothion demonstrated a decrease in their tendency to defend territories in artificial stream beds when compared with control fish. The effects were significant at a concentration of 0.1 mg/L (Symons 1973). Peterson (1976) demonstrated that there was no significant change in the temperature selection of juvenile Atlantic salmon exposed to fenitrothion concentrations up to 1.0 mg/L.

Beside water-borne contamination, it is possible that fish could be exposed to the toxicant through ingestion of contaminated food items. Symons (1973) determined, however, that young salmon would not voluntarily eat mealworms injected with fenitrothion. He also observed that fish which were force-fed contaminated food regurgitated it. The threshold for regurgitation was 2.5 µL of fenitrothion. A similar threshold was determined for brook trout feeding (Wildish and Lister 1973). Symons (1973) noted a cessation of feeding activity in fish exposed to water-borne fenitrothion at concentrations of 1.0 mg/L. Those combined observations indicate ingestion would not be an exposure route leading to acutely toxic effects. Wildish and Lister (1979) contaminated brook trout encapsulated commercial diet to a concentration of 10 µg/g (below regurgitation threshold) for four weeks and found no effects on growth or hierarchical position. In a similar feeding study, brooktrout fed a diet contaminated with 10 µg/g fenitrothion had no significant depression in brain AChE activity when compared with control fish (Wildish 1974).

In a series of intensive observations, Bull and McInerney (1974) determined that 2-h exposures to fenitrothion produced behavioural changes indicative of some degree of physiological impairment. These include: a reduction in feeding activity at 0.1 mg/L or greater; a reduction of social behaviours (displays, chases, displacements, flights, appeasements, and vacating) in the 0.03 mg/L range; an increase of cough frequency; and a cessation of locomotion in the 0.48-0.75 mg/L range.

4.11 FIELD STUDIES WITH FISH

Studies confined to the laboratory are useful, cost-effective screening and mechanism-determination tools. Unfortunately, they do not consider many of the larger environmental forces, or receiving environment characteristics which may modify the effect of a pesticide on fish. Factors such as meteorological conditions, amount of spray drift, amount of sunlight, microbial populations in receiving water, and even the existence of refugia can all affect the ultimate impact that the pesticide will have on fish. The only way that such factors can be accounted for is by measuring impacts after large-scale field applications. For the most part, field data have been obtained from studies associated with operational programs or large-scale field efficacy trials.

4.11.1 Direct Mortality

Most of the available information indicates that direct fish mortality, from operational sprays or experimental sprays at near operational dosage rates, does not occur frequently. There have been sporadic and generally poorly documented reports of fish mortality. Where mortalities have been observed, they have either not been definitively linked to fenitrothion spraying (almost an impossible task), or they have been attributed to greater than normal deposits in water, which may have been the result of navigational errors or of particular meteorological conditions.

Kingsbury (1973) documented a kill of approximately 30 brook trout in a 12 ha lake oversprayed three times in the same year during an operational spruce budworm spray program using a fenitrothion-in-oil formulation (dosage rate stated as 140-210 g a.i./ha). That mortality was associated with a rapid decline in angling success in the lake. In addition, two dead fish contained measured concentrations of fenitrothion of 0.77 and 0.19 µg/g. Likewise, Côté and Tetreault (1973) reported a large number of dead fish (141 fish in 24 h) found on a barrier net in a stream draining a spray block which had been treated, under operational conditions four days previously, with 140 g a.i./ha fenitrothion. The authors could not eliminate the possibility that a rapid change in water level, due to rain, caused impingement of small fish on the barrier net and subsequent mortality. There was, however, a progressive increase in the concentration of fenitrothion in dead fish collected from the barrier net which corresponded with the increase in numbers of fish collected.

Kingsbury (1977a), after a direct application of an aqueous emulsion of fenitrothion to a lake at 420 g a.i./ha (twice the normal forestry dosage rate), observed 100% mortality in caged white suckers (*Catostomus commersoni*) and bullheads (*Ictalurus melas*) within 24 h, and a 62% mortality in caged smallmouth bass (*Micropterus dolomieu*). No attempt was made to assess mortality in fish held in similar cages in unsprayed areas. Only one wild fish, a smallmouth bass, was found in a distressed condition. In that instance, caged fish mortality could have been due to the inability of fish to escape localized (surface) high concentrations of the pesticide or because of the additional stress that caging has on fish. That such stress may be a significant factor is evidenced by the fact that although Riche and Coady (1969) observed substantial mortality (up to 75%) in caged Atlantic salmon and brook trout parr and fry

in a stream draining blocks operationally treated with fenitrothion at 140-210 g a.i./ha (formulation unstated), similar mortalities were also observed in caged fish held in unsprayed streams.

During a Newfoundland forest spray program, Hatfield (1969) observed a small number of dead salmon parr (10 in 0.8 km of stream) two days after a fenitrothion spray at 280 g a.i./ha. However, he could not rule out fluctuating water levels as a possible cause of mortality. In the same study, he noted a similar number of dead salmon and trout after an operational spray of 140 g a.i./ha. Since the watercourses were protected by buffer zones where only phosphamidon was sprayed, he could not relate the observed mortalities directly to fenitrothion exposure.

Many reports, too numerous to detail individually, show no fish mortality after fenitrothion sprays at operational dosage rates. Some representative reports include those of Penney (1971a, 1971b), Carrow (1974), Marancik (1976a), Montreal Engineering Co. (1976, 1978), Kingsbury (1978a), Coady (1978), and Ernst and Julien (1984).

In some cases, where fenitrothion was applied at dosage rates substantially greater than those presently used in Canada, direct fish mortalities did not occur. In Scotland, when a 50:50 mixture of fenitrothion and butyl dioxtyl was used in a forest spray program (no indication of watercourse buffer zones) at a dosage rate of 300 g a.i./ha, there were no mortalities of caged brown trout (*Salmo trutta*) or Atlantic salmon (Morrison and Wells 1981). Prior to the registration of fenitrothion in Canada, Penney and MacDonald (1966) held Atlantic salmon parr and fry in streams where fenitrothion was directly applied at a dosage rate of 560 g a.i./ha (formulation unstated). The number of fish that died in the treatment stream cages was no different from the number in similar cages in untreated streams. In situations where fenitrothion (dosage rates unknown) was applied directly to rice paddies supporting populations of *Channa ophioccephalus*, no immediate fish mortalities were observed (Yunus and Soon 1971). That was not the case with pesticides such as endosulfan, BHC, azinphosethyl, dieldrin, and endrin applied at levels effective against the same pest.

4.11.2 Bioaccumulation

Residue concentrations in fish after operational use of fenitrothion are expectedly much more variable than those in fish after laboratory exposure (Table

4.21). A greater variation in deposit, water-quality characteristics, and microhabitat all function to provide a greater range of fenitrothion concentrations in fish exposed in the field.

Fenitrothion concentrations in fish after operational sprays are generally less than the maximum concentrations measured in laboratory-exposed fish. Bioconcentration factors in operational spray-exposed fish are generally one-tenth to one-hundredth those in laboratory-exposed fish. That is undoubtedly a function of the much lower concentrations of pesticides in water and their shorter existence in solution under field conditions. It appears from the data in Table 4.21, particularly Lockhart et al. (1973), that fish in lentic systems accumulate greater fenitrothion concentrations than fish in lotic systems. Lower initial deposits and more rapid disappearance of fenitrothion from lotic systems (Lockhart et al. 1977) could be the major cause. Some of the highest residues recorded (Lockhart et al. 1973) are from wild fish from a "stagnant" pool which was directly oversprayed. As previously stated, however, the residence time of fenitrothion in fish is quite short - less than one day in rainbow trout (Lockhart et al. 1973).

4.11.3 Acetylcholinesterase Inhibition

A number of investigators measured brain AChE depression in fish after operational sprays. The results obtained are not consistent. Lockhart et al. (1973) observed no differences in serum or brain AChE activity in rainbow trout caged in areas directly oversprayed with 280 g a.i./ha fenitrothion from fish caged in unsprayed areas. In the same study, changes in blood serum parameters such as alkaline phosphatase, inorganic phosphate, sodium, chloride, total protein, lactate dehydrogenase, and glutamic oxalacetic transaminase, which could be related to fenitrothion exposure, were undetected. In a similar study (Lockhart et al. 1977), no differences in the brain AChE activity of wild dace (*Chrosomus eos* and *C. neogaeus*), mudminnow (*Umbra limi*), darter, or stickleback were noted before and after spraying with fenitrothion at 280 g a.i./ha. Likewise, Fancey et al. (1987) measured no significant differences in brain AChE activity in wild Atlantic salmon or brooktrout collected from an area treated operationally with 210 g a.i./ha compared with fish in untreated areas. In that study, streams within spray blocks were protected by 400-m buffer zones.

With experimental fenitrothion formulations aerielly oversprayed at a dosage rate of 210 g a.i./ha, Ernst and Julien (1984) measured from 19-26% depression

in brain AChE of caged brooktrout compared with controls. However, due to the high variability of individual measurements (up to five pooled brains), the treatment samples were not significantly different from controls. While Marancik (1976a, 1976b) observed a minimum of only 6.8% depression of brain AChE in brook trout and 24% reduction in Northern creek chub (*Semotilus corporalis*) taken from areas sprayed operationally with fenitrothion at 210 g a.i./ha (no indication of watercourse buffer zones), the statistical significance of those depressions could not be determined. In a study which was not statistically valid, Morrison and Wells (1981) documented 30% reductions in brain AChE activity of brown trout due to operational spraying of fenitrothion at 300 g a.i./ha. While the ecological implications of sublethal AChE depression are as yet unclear, a 10% reduction is generally indicative of anti-cholinesterase exposure (Weiss 1961; Holland et al. 1967) and may have sublethal behavioural effects (Jarvinen et al. 1983). Depressions of greater than 50% of control values have been associated with some degree of lethality (Macek et al. 1972; Jarvinen et al. 1983).

In the one study attempting to measure developmental effects of operational spraying on fish, Hiraoka and Okuda (1984b) exposed medaka embryos in streams (with unstated buffer zones) draining operational forestry blocks sprayed with fenitrothion at an estimated dosage rate of 1.84 kg/ha. The developing embryos were removed from the exposure streams one week after spraying and reared to adulthood in clean water in the laboratory. Exposed fish had an incidence of vertebral centrum deformities of 34-66% while in control fish (exposed in unsprayed streams), the incidence of similar damage was 11%.

4.11.4 Population Effects

The greatest potential for long-term influence of fenitrothion stems from possible effects on individuals and populations from reductions in food availability. As indicated in the previous discussion, there can be major changes in the availability of food items. However, surprisingly few investigators attempted to measure impacts of food-resource changes on fish populations induced by fenitrothion spraying.

It has been demonstrated that fish may initially increase their intake of food organisms directly after spraying due to the increased number of insecticide-affected invertebrates within the water column. Ernst and Julien (1984) observed that, compared with controls, brook trout in streams directly oversprayed with fenitrothion at 210 g a.i./ha had

Table 4.21 Concentration of fenitrothion in fish after aerial spray programs.

Fish species	Application rate (g a.i./ha)	Measured water concent. maximum (mg/L)	Tissue	Maximum tissue concentration (µg/g)	Bio- concentration factor*	Reference
Brown trout (<i>Salmo trutta</i>)	300	18.8 (lotic)	stomach & contents skin muscle liver	0.24 0.16 0.13 0.08	12.8 8.5 6.9 4.2	Morrison and Wells 1981
Brook trout (<i>Salvelinus fontinalis</i>)	210		whole fish	0.58		Marancik 1976a
Creek chub (<i>Semotilus atromaculatus</i>)		(lotic)	whole fish	0.13		
Blacknose dace (<i>Rhinichthys atratulus</i>)			whole fish	0.38		
White sucker (<i>Catostomus commersoni</i>)	140	6.62	whole fish	0.48	72.5	Marancik 1976a
Brook trout		(lotic)	whole fish	1.21	182.8	
Lake chub (<i>Couesius plumbeus</i>)			whole fish	0.33	50.0	
White sucker	420	21.6	whole fish	1.01	46.8	Kingsbury 1977a
Fallfish (<i>Semotilus corporalis</i>)		(lentic)	whole fish	0.76	35.2	
Brown bullhead (<i>Ictalurus nebulosus</i>)	420	21.6 (lentic)	whole fish	0.44	20.4	Kingsbury 1977a
Smallmouth bass (<i>Micropterus dolomieu</i>)	420	21.6 (lentic)	whole fish	0.34	15.7	Kingsbury 1977a
Rainbow trout (<i>Salmo gairdneri</i>)	280	(lotic)	whole fish	0.63		Lockhart et al. 1973
Stickleback (<i>Culea inconstans</i>)			whole fish	4.97		
Dace (<i>Chrosomus eos</i>)				13.7		

(continued)

Table 4.21 (continued)

Fish species	Application rate (g a.i./ha)	Measured water concent. maximum (mg/L)	Tissue	Maximum tissue concentration (µg/g)	Bio- concentration factor*	Reference
Atlantic salmon	140	(lotic)	whole fish	2.70		Hatfield and Riche 1970 Malis and Muir 1984
Fathead minnow (<i>Pimephales promelas</i>)	165	70 (lentic)	whole fish	4.92	70	
Brook trout	280	51.6 (lentic)	fat	0.21	4.1	Holmes et al. 1984
			muscle	0.16	3.1	
			intestine	0.12	2.3	
			liver	0.28	5.4	
			ovaries	0.17	3.3	
Lake trout (<i>Salvelinus namaycush</i>)			viscera	0.18	3.5	
			fat	0.66	12.8	
			muscle	0.31	6.0	
			intestine	0.31	6.0	
			liver	0.97	18.8	
Mud minnow (<i>Umbra limi</i>)	280	701 (lentic)	ovaries	0.48	9.3	Lockhart et al. 1973
			whole fish	1.32	1.8	
Dace (<i>Chrosomus eos</i>)	280	701	whole fish	1.58	2.2	
Dace (<i>Chrosomus neogaeus</i>)	280		whole fish	0.57	0.8	Lockhart et al. 1973
Stickleback			whole fish	4.28	6.1	
Mud minnow	180	1.32 (lotic)	whole fish	0.1	75.8	Lockhart et al. 1977
			whole fish	0.47	356.1	
Dace (<i>C. eos</i>)			whole fish	0.16	121.2	
Dace (<i>C. neogaeus</i>)			whole fish	0.61	462.1	
Stickleback			whole fish			

* The bioconcentration factors were derived by dividing the maximum measured tissue concentration by the maximum measured water concentration. This may not be the method used by the original authors to derive bioconcentration factors.

greater numbers of food items and a greater total volume of food in their guts immediately after spray. Gut contents consisted primarily of knocked down spruce budworm larvae as well as stonefly (Plecoptera) and blackfly (Diptera) larvae. Kingsbury (1978a, 1978b) made a similar observation in brooktrout from lakes and streams directly sprayed with fenitrothion in an oil formulation at 210 g a.i./ha; increased food was primarily baetid mayfly nymphs (Ephemeroptera), and caddisfly larvae (Trichoptera) in lakes and immature mayfly, stonefly, blackfly, and midge larvae (Diptera) in streams.

By comparing numbers of fish captured from streams in areas operationally sprayed for spruce budworm control with fish from streams not sprayed, many early investigators detected no population density effects on native Atlantic salmon or brook trout (Penney and MacDonald 1966; MacDonald and Penney 1968, 1969; Penney 1971b). How rigorously those early studies were conducted cannot be assessed because the methods employed were not fully described and no statement of statistical confidence was provided.

In a study limited by sample size, Symons and Harding (1974) attempted to determine biomass changes in stream fish which could be related to fenitrothion treatments. Five streams within five spray blocks treated operationally with fenitrothion at dosage rates from 140-210 g a.i./ha and two control streams were sampled for fish density and weight. There was no statement of the presence or absence of buffer zones. They observed that in the two control streams total fish biomass increased at a rate greater than in four of the five sprayed streams. They also observed that growth rates of brook trout fry were reduced in sprayed areas compared with those in control streams. However, the spray regime did not affect the growth of under-yearling dace (*Rhinichthys atratulus* and *Chrosomus eos*). The major short-comings of the study were that fish movement (emigration and immigration) was uncontrolled, and the numbers of treatments and samples were insufficient to obtain statistical significance (growth rate estimates were sometimes based on single fish). As a separate part of the study, investigators trickled a fenitrothion formulation into a brook, producing water concentrations ranging from 0.05-0.5 mg/L for a 24-h period. That was sufficient to cause substantial insect mortality as well as a 50% reduction of total fish biomass compared with controls, not due to fish mortality, within one month of treatment. Comparable growth rates (comparing two trout) occurred in treated and untreated areas.

In a contract study commissioned by the Province of New Brunswick (Montreal Engineering Co. 1978), an attempt was made to further define the effect of food reduction on fish (Atlantic salmon and brook trout) growth. Ten streams were monitored in an area treated twice with 210 g a.i./ha of fenitrothion. There was no indication of the presence of watercourse buffer zones. Those treatments had measurable impacts on both macro-invertebrate drift and benthos, but recoveries were evident within two months of treatment. The data for fish populations among streams were highly variable (a reflection of great differences in habitat suitability). For that reason, smaller growth rates (actually negative growth rates in some instances) in treated areas were not significant when compared with controls. There were two flaws in the study: loss of fish to emigration could not be accounted for; and samples of water obtained from control areas had concentrations of fenitrothion, of unknown source, similar to concentrations measured in samples from treated areas (0-25 µg/L). The investigators present the case that fenitrothion has no effects on fish as evidenced by the fact that the population of salmonids in treatment areas, many of which had a history of spraying in the previous nine years, was as great, if not greater, than populations in control areas which had no previous spraying history. That logic is flawed by the fact that obvious site differences (i.e. the areas differed in their susceptibility to spruce budworm infestation) were not accounted for in normalizing population data. Many of the same streams were examined two years later after they were subjected to a similar (2 x 210 g a.i./ha) fenitrothion treatment (Montreal Engineering Co. 1980). The investigators were unable to detect any difference in mean lengths of brooktrout in sprayed areas, compared with unsprayed areas. Atlantic salmon were consistently, though not significantly, longer in unsprayed streams. The mean weight of salmon from the unsprayed area was greater than that of salmon from the sprayed area. Because of a greater growth rate of fish in the sprayed area however, that difference decreased as the summer proceeded and was negligible within two months of spraying. The authors attributed that observation to the fact that water temperatures were less conducive to growth in the control stream. Population densities of fish of all species were similar in sprayed and unsprayed areas. Caution is warranted in interpreting those results because all water samples from the sprayed area had a very low level of contamination (0.02-0.64 µg/L) and no invertebrate samples were taken, therefore there was no indication of food-source impact.

4.12 INFORMATION DEFICIENCIES

Fenitrothion residues may persist longer and cause aquatic invertebrate depletions of significant magnitude in small lentic habitats, in particular bog ponds. The implications of such depletions for energy flow and higher faunal productivity in those habitats need to be explored. Suitable watercourse buffer zones need to be devised and implemented for small lentic water bodies.

There is a need for further investigation of the relationships among pesticide exposure, fish tissue residue concentrations, AChE depression and physiological impairment in fish.

Developmental abnormalities have been observed in fish embryos exposed to fenitrothion in the laboratory and under high dosage rate operational field conditions in Japan. There is a need for studies to determine if embryonic development of native fish could be affected by Canadian fenitrothion use patterns.

4.13 LITERATURE CITED

- Ahern, A. and J. Leclerc. 1981. Comparative study of the impact of Matacil and fenitrothion on benthic populations in running water. Bio-Conseil Inc., Québec. 94 p.
- Allen, J. D. 1984. Hypothesis testing in ecological studies of aquatic insects. p. 484-507. In V. H. Resh and D. M. Rosenberg (ed.) The ecology of aquatic insects. Praeger Publishers, New York. 625 p.
- Anjum, F. and S. S. H. Quadri. 1986. *In vivo* metabolism of fenitrothion (O,O-dimethyl-O-(4-nitro-m-tolyl) phosphorothioate) in fresh water teleost (*Tilapia mossambica*). Bull. Environ. Contam. Toxicol. 36: 140-145.
- Baarschers, W. H., J. Elvish, and S. P. Ryan. 1983. Adsorption of fenitrothion and 3-methyl-4-nitrophenol on soils and sediment. Bull. Environ. Contam. Toxicol. 30: 621-627.
- Baarschers, W. H. and H. S. Heitland. 1986. Biodegradation of fenitrothion and fenitrooxon by the fungus *Trichoderma viride*. J. Agr. Food Chem. 34: 707.
- Banks, D. B. 1973. The influence of aerially applied fenitrothion on aquatic invertebrates in Cove Stream, New Brunswick. M.Sc. thesis, Acadia University, Nova Scotia. 86p.
- Banks, D. B. and H. A. Hall. 1972. Summary report on the effects of the 1972 forest spray program in New Brunswick on aquatic invertebrates. Env. Prot. Serv., Env. Can. Surv. Rep. 77 p.
- Berkane, K., G. E. Cassie, and V. N. Mallet. 1977. The use of amberlite XAD-2 resin for the quantitative recovery of fenitrothion (organophosphorus insecticide) from water - a preservation technique. J. Chromatogr. 139: 386-390.
- Buckner, C. H. and B. B. McLeod. 1977. Ecological impact studies of experimental and operational spruce budworm (*Choristoneura fumiferana* Clemens) control programs on selected non-target organisms in Québec, 1976. Can. For. Serv., For. Pest. Manage. Inst. Rep. CC-X-137. 81 p.
- Buckner, C. H., B. B. McLeod, and R. G. Lidstone. 1976. Environmental impact studies of spruce budworm (*Choristoneura fumiferana* Clemens) control programs in New Brunswick in 1976. Can. For. Serv., Chem. Control Res. Inst. Rep. CC-X-135. 140 p.
- Bull, C. J. and J. E. McInerney. 1974. Behaviour of juvenile Coho salmon, *Oncorhynchus kisutch*, exposed to Sumithion fenitrothion, an organophosphate insecticide. J. Fish. Res. Board Can. 31: 1867-1872.
- Cahill, M. 1987. Buffer zone regulations applied to forest insect control operations in Newfoundland and Labrador: An evolutionary perspective p.2-3. In Buffer zones: Their application to forest insect control operations. Proc. Buffer Zone Workshop, Eastern Spruce Budworm Council's Environmental Committee, Québec City 16-17 April, 1986. 124 p. ISBN 0-662-15732-X.
- Carrow, J. R. (ed). 1974. Aerial spraying operations against blackheaded budworm on Vancouver Island, 1973. Env. Can. Inf. Rep. Bc-X-101. 56 p.
- Cassista, A. and V. N. Mallet. 1984. Simultaneous GLC determination of fenitrothion, aminocarb, and their corresponding hydrolysis products in water. Chromatogr. 18: 305.
- Chiou, C. T., V. H. Freed, D. W. Schmedding, and R. L. Kohnert. 1977. Partition coefficient and bioaccumulation of selected organic chemicals. Env. Sci. Technol. 11: 475.
- Coady, L. W. 1978. Immediate chemical and toxicological influences of aerially applied fenitrothion and aminocarb in selected Newfoundland streams. Env. Can. Surv. Rep. EPS 5-AR-78-1. 83 p.
- Côté, Y. and B. Tetreault. 1973. Short-term effects of aerial fenitrothion spraying on juvenile salmon (*Salmo salar* L.) in Anticosti Island, Québec. Atl. Salmon J. 3: 18-21.

- Dimond, J. B. and S. A. Malcolm. 1971. Accothion and aquatic insects: Monitoring of stream populations. p. 60-68. In Environmental studies of Accothion for spruce budworm control. A cooperative study by the State of Maine, U.S. Dept. Agriculture and U.S. Dept. Interior.
- Doe, K. G., W. R. Ernst, W. R. Parker, G. Julien, and P. Hennigar. 1988. Influence of pH on the acute lethality of fenitrothion 2,4-D and aminocarb and some pH altered sublethal effects of aminocarb to rainbow trout (*Salmo gairdneri*). Can. J. Fish. Aquat. Sci. 45: 287-293.
- Dostie, R., and G. Parent. 1986. Comportement du fénitrothion (Sumithion) en milieu lotique à l'extérieur des aires traitées. Ministère de l'Énergie et des Ressources, Québec. 14 p.
- Dostie, R., P. M. Marotte, and G. Parent. 1985. Résidus de fénitrothion et d'aminocarbe dans l'eau et le feuillage. Ministère de l'Énergie et des Ressources, Québec. 36 p.
- Dostie, R., P. M. Marotte, and S. Delisle. 1987. Concentrations de fénitrothion dans l'eau de rivières à saumons situées à l'intérieur des aires traitées contre la tordeuse des bourgeons de l'épinette en 1986. Service des études environnementales, Ministère de l'Énergie et des Ressources, Québec, 20 p.
- Duangasawadi, M. and J. F. Klavervkamp. 1979. Acephate and fenitrothion toxicity in rainbow trout, effects of temperature stress and investigations on the sites of action. In J. B. Wheeler (ed.). ASTM Spec. Tech. Publ. No. 667. Philadelphia. p. 35-51.
- Eidt, D. C. 1975a. The effect of fenitrothion from large-scale spraying on benthos in New Brunswick head water streams. Can. Entomol. 107: 743-760.
- Eidt, D. C. 1975b. Effects of fenitrothion from 1974 forest spraying on benthos of the Nashwaak project study streams. Can. For. Serv., Inf. Rep. M-X-49. 13 p.
- Eidt, D. C. 1977. Effects of fenitrothion on benthos in the Nashwaak project study streams in 1976. Can. For. Serv., Inf. Rep. M-X-70. 22 p.
- Eidt, D. C. 1978. Toxicity of fenitrothion to insects in a woodland stream. Can. For. Serv. Inf. Rep. M-X-86: 13 p.
- Eidt, D. C. 1981. Recovery of aquatic arthropod populations in a woodland stream after depletion by fenitrothion treatment. Can. Entomol. 113: 303-313.
- Eidt, D. C. and K. M. S. Sundaram. 1974. Fenitrothion residues in trout streams from large-scale forest spraying. Env. Can., Can. For. Serv. File Report.
- Eidt, D. C. and K. M. S. Sundaram. 1975. The insecticide fenitrothion in headwaters streams from large-scale forest spraying. Can. Entomol. 107: 735-742.
- Eidt, D. C. and C. A. Weaver. 1984a. Influence of site and fenitrothion contamination on vertical distribution of drift arthropods in a woodland stream. Can. Entomol. 116: 1425-1430.
- Eidt, D. C. and C. A. Weaver. 1984b. The fenitrothion and aminocarb content and the distribution of the aquatic plant *Ceratophyllum demersum* relative to forest spraying in New Brunswick and Nova Scotia. Naturalist Can. 111: 235-239.
- Eidt, D. C., A. J. Sosiak, and V. N. Mallet. 1984. Partitioning and short-term persistence of fenitrothion in New Brunswick (Canada) headwater streams. Arch. Environ. Contam. Toxicol. 13: 43-52.
- Elliot, J. M. 1977. Some methods for the statistical analysis of samples of benthic invertebrates. Freshwater Biological Association, Sci. Pub. No. 25. 160 p.
- Elson, P. F., J. W. Saunders, and V. Zitko. 1972. Impact of forest-based industries on freshwater-dependent fish resources in New Brunswick. Fish. Res. Board Can. Tech. Rep. 325. 26.
- Ernst, W. R. and G. R. Julien. 1984. Effects on brook trout (*Salvelinus fontinalis*) feeding behaviour and brain cholinesterase activity of experimental fenitrothion and aminocarb formulations containing Triton X-100 and Cyclosol R 63. Env. Can. Surv. Rep. EPS-5-AR-84-9. 28 p.

- Ernst, W., G. Julien, K. Doe, and R. Parker. 1980. Environmental investigations of the 1980 spruce budworm spray program in New Brunswick. Env. Can. Surv. Rep. EPS-5-AR-81-3. 46 p.
- Ernst, W.R., K.G. Doe, and P.A. Hennigar. (in prep.) A comparison of the aquatic toxicity of fenitrothion flowable and fenitrothion emulsion formulations. Environment Canada, Dartmouth, Nova Scotia.
- Fairchild, W. (in prep.) Perturbation of the aquatic invertebrate community of acidic bog ponds by the insecticide fenitrothion. PhD thesis, University of New Brunswick, Fredericton, New Brunswick.
- Fairchild, W. L. and D. C. Eidt. 1988. Perturbation of the aquatic invertebrate community of bog ponds by fenitrothion and aminocarb. p. 44 In W. A. Sexsmith (ed.). Environmental Monitoring of Forestry Control Operations (EMOFCO). New Brunswick Dept. of Municipal Affairs and Environment. 70 p.
- Fairchild, W. L., D. C. Eidt, and C. A. Weaver. 1987. Effects of fenitrothion insecticide on inhabitants of leaves of the pitcher plant, *Sarracenia purpurea* L. Can. Entomol. 119: 647-652.
- Fancey, L. L., J. F. Payne, M. A. Barnes, and R. N. McCubbin. 1987. Acetylcholinesterase enzyme activity in fish sampled after the 1985 Newfoundland forest spray program for eastern hemlock looper using fenitrothion. Fish. Aquat. Sci. Can. Tech. Rep. 1579. 10 p.
- Fisher, S. W. 1985. Effects of pH upon the environmental fate of ^{14}C -fenitrothion in an aquatic microcosm. Ecotoxicol. Environ. Saf. 10: 53-62.
- Flannagan, J. F. 1973. Field and laboratory studies of the effect of exposure to fenitrothion on freshwater aquatic invertebrates. Manit. Entomol. 7: 15-25.
- Foisy, L. et al. 1977. Parcs Nationaux. Parcs Canada. 137 p.
- Forbes, N. A., B. P. Wilson, R. Greenhalgh, and W. P. Cochrane. 1975. Confirmation of organophosphorus insecticides by chemical reduction. Bull. Environ. Contam. Toxicol. 13: 141.
- Gaboury, G. 1980. Surveillance environnementale des pulvérisations aériennes 1979 contre la tordeuse des bourgeons de l'épinette au Québec (rapport préliminaire). Ministère de l'Énergie et des Ressources, Québec. 20 p.
- Gaboury, G., P. M. Marotte, R. Morin, and M. Pelletier. 1981. Compilation bibliographique concernant la toxicité et les effets environnementaux du fenitrothion, de l'aminocarb et du *Bacillus thuringiensis* Berliner. Ministère de l'Énergie et des Ressources, Québec. Rapport interne EP-81-02. 380 p.
- Green, R. H. 1979. Sampling design and statistical methods for environmental biologists. John Wiley and Sons, New York. 257 p.
- Greenhalgh, R. and W. D. Marshall. 1976. Photolysis of Fenitrothion. J. Agr. Food Chem. 24: 708.
- Greenhalgh, R., K. L. Dhawan, and P. Weinberger. 1980. Hydrolysis of fenitrothion in model and natural aquatic systems. J. Agr. Food. Chem. 28: 102-105.
- Grift, N. and W. L. Lockhart. 1974. Gas-liquid chromatographic determination of fenitrothion in fish, water, and sediment. J. Assoc. Off. Anal. Chem. 57: 1282-1284.
- Haché, P., R. Marquette, G. Volpé, and V. N. Mallet. 1981. Fast cleanup of difficult substrates for determination of fenitrothion and some derivatives. J. Assoc. Off. Anal. Chem. 64: 1470-1473.
- Hall, H. A. and P. B. Eaton. 1973. Summary report on the effects of the 1973 forest spray program in New Brunswick on aquatic invertebrates. Forest Pest Control Forum, 1973, Can. For. Serv., Ottawa. Appendix 4.
- Hasegawa, J., M. Yasuno, K. Saito, Y. Nakamura, S. Hatakeyama, and O. H. Sato. 1982. Impact of temephos and fenitrothion on aquatic invertebrates in a stream of Mount Tsukuba, Japan. Jap. J. Sanit. Zool. 33: 363-368.
- Hashimoto, Y., E. Okubo, T. Ito, M. Yamaguchi, and S. Tanaka. 1982. Changes in susceptibility of carp to several pesticides with growth. J. Pestic. Sci. 7: 457-462.

- Hatfield, C. T. 1969. Effects of aerial spraying with Sumithion and Phosphamidon on salmon brooks in western and central Newfoundland. Res. Dev. Branch, Dept. Fish. Can. 32 p.
- Hatfield, C. T. and J. M. Anderson. 1972. Effects of two insecticides on the vulnerability of Atlantic salmon (*Salmo salar*) parr to brook trout (*Salvelinus fontinalis*). J. Fish. Res. Board Can. 29: 27-29.
- Hatfield, C. T. and P. H. Johansen. 1972. Effects of four insecticides on the ability of Atlantic salmon (*Salmo salar*) to learn and retain a simple conditioned response. J. Fish. Res. Board Can. 29: 315-321.
- Hatfield, C. T. and L. G. Riche. 1970. Effects of aerial Sumithion spraying on juvenile Atlantic salmon (*Salmo salar* L.) and brook trout (*Salvelinus fontinalis* Mitchell) in Newfoundland. Bull. Environ. Contam. Toxicol. 5: 440-442.
- Hiraoka, Y. and H. Okuda. 1984a. Time course study of occurrence of anomalies in medakas *Oryzias latipes* centrum by cadmium or fenitrothion emulsion. Bull. Environ. Contam. Toxicol. 32: 693-697.
- Hiraoka, Y. and H. Okuda. 1984b. A tentative assessment of water pollution by the medaka, *Oryzias latipes*, egg stationing method aerial application of fenitrothion emulsion. Environ. Res. 34: 262-267.
- Holland, H. T., D. L. Coppage, and P. A. Butler. 1967. Use of fish brain cholinesterase to monitor pollution by organophosphorus pesticides. Bull. Environ. Contam. Toxicol. 2: 156-162.
- Holmes, S. B. 1979. Aquatic impact studies of a spruce budworm control program in the lower St. Lawrence region of Québec in 1978. Can. For. Serv., For. Pest. Manage. Inst. Rep. FPM-X-26. 34 p.
- Holmes, S. B. 1982a. Aquatic impact studies on experimental aqueous formulations of spruce budworm control agents containing Triton X-100 to rainbow trout. Can. For. Serv., For. Pest. Manage. Inst. File Rep. 38. 21 p.
- Holmes, S. B. 1982b. Lethal toxicity of experimental aqueous formulations of spruce budworm control agents containing Triton X-100 to rainbow trout. Can. For. Serv., For. Pest. Manage. Inst. File Rep. 42. 7 p.
- Holmes, S. B., P. D. Kingsbury, G. Mamarbachi, and P. Mathieu. 1984. Distribution of fenitrothion residues in brook trout (*Salvelinus fontinalis*) and lake trout (*Salvelinus namaycush*) tissues following aerial applications to Lac Ste Anne, Québec. Bull. Environ. Contam. Toxicol. 33: 468-475.
- Hurlbert, S. H. 1984. Pseudoreplication and the design of ecological field experiments. Ecol. Monogr. 54: 187-211.
- Hynes, H. B. N., D. D. Williams, and N. E. Williams. 1976. Distribution of the benthos within the substrates of a Welsh mountain stream. Oikos 27: 307-310.
- Jarvinen, A. W., B. R. Nordling, and M. E. Henry. 1983. Chronic toxicity of Dursban (chlorpyrifos) to the fathead minnow (*Pimephales promelas*) and the resultant acetylcholinesterase inhibition. Ecotoxicol. Environ. Safety 7: 423-434.
- Johnson, W. W. and M. T. Findlay. 1980. Handbook of acute toxicity of chemicals to fish and aquatic invertebrates. U.S. Dept. Interior, Fish. Wildl. Serv., Res. Pub. 137. 96 p.
- Kamaldeep, K. and H. S. Toor. 1977. Toxicity of pesticides to embryonic stages of *Cyprinus carpio communis* Linn. Indian J. Exp. Biol. 15: 193.
- Kanazawa, J. 1975. Uptake and excretion of organophosphorus and carbamate insecticides by fresh water fish, Mutsugo, *Pseudorasbora parva*. Bull. Environ. Contam. Toxicol. 14: 346-352.
- Kanazawa, J. 1981. Bioconcentration of pesticides in freshwater fish correlated with chemical properties and acute toxicity. Pestic. Sci. 12: 417.
- Kapur, K. and S. Virk. 1983. Effect of BHC and Sumithion on ovarian recrudescence in common carp, *Cyprinus carpio*. Indian J. Ecol. 10: 182-187.

- Kapur, K., K. Kamaldeep, and H. S. Toor. 1978. The effect of fenitrothion on reproduction of a teleost fish, *Cyprinus carpio communis*, a biochemical study. Bull. Environ. Contam. Toxicol. 20: 438-442.
- Kikucki, R., Y. Takimoto, H. Yamada, and J. Miyamoto. 1984. Effects of fenitrothion on growth of a green alga, *Chlorella vulgaris*. Nippon Noyaku Gakkaishi. 9: 325-329.
- Kingsbury, P. D. 1973. Investigation of fish mortalities in and adjacent to areas of Québec treated with the insecticide fenitrothion in 1973. Can. For. Serv., Chem. Control Res. Inst. Rep. CC-X-58. 19 p.
- Kingsbury, P. D. 1977a. Fenitrothion in a lake ecosystem. Can. For. Serv., Chem. Control Res. Inst. Rep. CC-X-146. 123 p.
- Kingsbury, P. D. 1977. Environmental monitoring by CCRI in Lac Ste. Anne, Québec spruce budworm spray block 305, 1977. Can. For. Serv., Chem. Cont. Res. Inst. File Rep. No. 77.
- Kingsbury, P. D. 1978a. Aquatic impact studies by Forest Pest Management Institute in Québec spruce budworm spray block 305, 1977. Can. For. Serv., For. Pest. Manage. Inst. Rep. FPM-X-12. 85 p.
- Kingsbury, P. D. 1978b. A study of the distribution, persistence and biological effects of fenitrothion applied to a small lake in an oil formulation. Can. For. Serv., For. Pest Manage. Inst. Inf. Rep. FPM-X-13. 85 p.
- Kingsbury, P. D. and R. Sarrazin. 1975. Aquatic fauna, p. 80-100. In C. H. Buckner and R. Sarrazin (eds.). Studies of the environmental impact of the 1974 spruce budworm control operation in Québec. Can. For. Serv., Chem. Control Res. Inst. Rep. CC-X-93. 106 p.
- Kingsbury, P. D. and J. G. Trial. 1987. Buffer zones: Their application to forest insect control operations. Proc. Buffer Zone Workshop, Eastern Spruce Budworm Council's Environmental Committee, Québec City, 16-17 April, 1986. 124 p.
- Klaverkamp, J. F., M. Duangsawasdi, W. A. MacDonald, and H. S. Majewski. 1977. An evaluation of fenitrothion toxicity in 4 life stages of rainbow trout, *Salmo gairdneri*. p. 231-240. In F. L. Mayer and J. L. Hamelink (ed.). Aquatic toxicology and hazard evaluation. ASTM Spec. Tech. Publ. No. 634, Philadelphia.
- Klaverkamp, J. F. and B. R. Hobden. 1980. Brain acetylcholinesterase inhibition and hepatic activation of acephate and fenitrothion in rainbow trout, *Salmo gairdneri*. Can. J. Fish Aquat. Sci. 37: 1450-1453.
- Kleiner, C. F., R. L. Anderson, and D. K. Tanner. 1984. Toxicity of fenitrothion to fathead minnows (*Pimephales promelas*) and alternate exposure studies with fenitrothion and endosulphan. Arch. Environ. Contam. Toxicol. 13: 573-578.
- Kodama, T. and S. Kowatsuka. 1980. Factors for the persistence of parathion, methyl parathion and fenitrothion in sea water. Nippon Noyaku Gakkaishi 5(3): 351-356.
- Koundinya, P. R. and R. Ramamurthi. 1978. Effect of Sumithion (fenitrothion) on some selected enzyme systems in the fish *Tilapia mossambica* (Peters). Indian J. Exp. Biol. 16: 809-811.
- Koundinya, P. R. and R. Ramamurthi. 1979a. Tissue respiration in *Tilapia mossambica* exposed to lethal (LC50) concentration of Sumithion and Sevin. Indian J. Env. Health 20: 426-428.
- Koundinya, P. R. and R. Ramamurthi. 1979b. Effect of organophosphorus pesticide Sumithion (fenitrothion) on some aspects of carbohydrate metabolism in a fresh water fish, *Sarotherodon mossambicus*. Experientia (Basel). 35: 1632-1633.
- Koundinya, P. R. and R. Ramamurthi. 1982. Effect of organophosphorus pesticide Sumithion fenitrothion on alkaline phosphatase activity of fresh water teleost *Sarotherodon mossambicus*. Curr. Sci. (Bangalore) 51: 191.

- Koundinya, P. R., R. Ranganatha, and R. Ramamurthi. 1978. Tissue respiration in *Tilapia mossambica* exposed to lethal (LC50) concentration of Sumithion and Sevin. Indian J. Env. Health 20 (4): 426-428.
- Kovacicova, J., V. Batora, and S. Truchlik. 1973. Hydrolysis rate and *in vitro* anticholinesterase activity of fenitrothion and S-methyl fenitrothion. Pestic. Sci. 4: 759-763.
- Lakshminarayana, J. S. S. and H. Bourque. 1980. Absorption of fenitrothion by plankton and benthic algae. Bull. Environ. Contam. Toxicol. 24: 389-396.
- Leonhard, S. L. 1974. Uptake of fenitrothion by caged crayfish *Orconectes virilis* in Pine Creek, Manitoba, 1973. Manit. Entomol. 8: 16-18.
- Lockhart, W. L., D. A. Metner, and N. Grift. 1973. Biochemical and residue studies on rainbow trout, *Salmo gairdneri*, following field and laboratory exposures to fenitrothion. Manit. Entomol. 7: 26-36.
- Lockhart, W. L., J. F. Flannagan, R. P. Moody, P. Weinberger, and R. Greenhalgh. 1977. Fenitrothion monitoring in southern Manitoba. NRCC Publ. No. NRCC 16073. Ottawa. p. 233-252.
- Lockhart, W. L., D. A. Metner, A. P. Blouw, and D. C. G. Muir. 1982. Prediction of biological availability of organic chemical pollutants to aquatic animals and plants. ASTM. Spec. Tech. Publ.: 766. 259-272.
- Lord, D. A., R. A. F. Matheson, L. Stuart, J. J. Swiss, and P. G. Wells. 1978. Environmental monitoring of the 1978 spruce budworm program in New Brunswick, Canada. Env. Can., Surv. Rep. EPS-5-AR-78-3. 161 p.
- MacDonald, J. R. and G. H. Penney. 1968. Report on the effects of the 1968 New Brunswick aerial forest spraying on juvenile salmon and their food organisms. Dept. Fish. Can., Resource Dev. Branch. 27 p.
- MacDonald, J. R. and G. H. Penney. 1969. Preliminary report on the effects of the 1969 New Brunswick forest spraying on juvenile salmon and their food organisms. Dept. Fish. For. Can., Resource Dev. Branch. 17 p.
- Macek, K. J., D. F. Walsh, J. W. Hogan, and P. P. Holz. 1972. Toxicity of the insecticide dursban to fish and aquatic invertebrates in ponds. Trans. Amer. Fish. Soc. 101: 420-427.
- Maguire, R. J. and E. J. Hale. 1980. Fenitrothion sprayed on a pond: kinetics of its distribution and transformation in water and sediment. J. Agric. Food. Chem. 28: 372-378.
- Major, L. and R. Dostie. 1983. Surveillance des pulvérisations aériennes d'insecticide contre le diprion de swaine au Québec en 1983 résidus de fenitrothion dans l'eau et les bleuets évaluation du depot. Ministère de l'Énergie et des Ressources.
- Malis, G. P. and D. C. G. Muir. 1984. Fate of fenitrothion in shaded and unshaded ponds. p. 277-296. In W. Y. Garner and J. Harvey Jr. [eds.]. Chemical and biological controls in forestry. A.C.S. Symp. Ser. 238. 397 p.
- Mallet, V. N. 1978. Report on the determination of fenitrothion in shellfish from the New Brunswick coastline. Unpub. Rep. U.de Moncton. Sept. 1978.
- Mallet, V. N. 1980. A chemical residue survey in relation to the 1980 spruce budworm spray program in New Brunswick. Report No. 80-CF-18, Environment New Brunswick, Fredericton, NB. 55 p.
- Mallet, V. N., P. E. Belliveau, and R. W. Frei. 1975. *In situ* fluorescence spectroscopy of pesticides and other organic pollutants. Residue Rev. 59: 51.
- Mallet, V. N., G. L. Brun, R. N. Macdonald, and K. Berkane. 1978. A comparative study of the use of XAD-2 resin and the conventional serial solvent extraction procedure for the analysis of fenitrothion and some derivatives in water preservation techniques. J. Chromatogr. 160: 81-88.

- Mallet, V. N. and G. Volpé. 1982. A chemical residue survey in relation to the 1980 spruce budworm spray program in New Brunswick (Canada). J. Environ. Sci. Health B17: 71-736.
- Mallet, V. N. and A. Cassista. 1984. Fenitrothion residue survey in relation to the 1981 spruce budworm spray program in New Brunswick, Canada. Bull. Environ. Contam. Toxicol. 32: 65-74.
- Mamarbachi, G., R. Dostie, and P. M. Marotte. 1985. Persistence des résidus d'aminocarbe et de fenitrothion dans les eaux naturelles en milieu forestier. Ministère de l'Énergie et des Ressources, Québec. 19 p.
- Mamarbachi, G., P. M. Marotte, and R. Dostie. 1987. Résidues de fenitrothion, d'aminocarbe et de leurs métabolites dans des organismes exposés aux pulvérisations aériennes contre la tordeuse des bourgeons de l'épinette au Québec en 1983 et 1984. Service des études environnementales, Ministère de l'Énergie et des Ressources, Québec. 26 p.
- Mandal, P. K. and A. K. Kulshrestha. 1980. Histopathological changes induced by the sublethal Sumithion in *Clarias batrachus*. Indian J. Exp. Biol. 18: 547-552.
- Marancik, J. 1976a. Monitoring of fish—1975 pesticide application for spruce budworm control. U.S. Dept. Interior, Fish Wildl. Serv., Fishery Services Division, Special Report. 38p.
- Marancik, J. 1976b. Effects of insecticides used for spruce budworm control in 1975 on fish, p. 11-34. In U.S. For. Serv., 1975 Cooperative pilot control project of Dylox, Matabacil, and Sumithion for spruce budworm control in Maine. U.S. For. Serv., Upper Darby, PA.
- Marotte, P. M. 1981. Évaluation des impacts sur les invertébrés aquatiques d'eaux courantes. Ministère de l'Énergie et des Ressources, Québec. 29 p.
- Marshall, W. K. and J. R. Roberts. 1977. Simulation modeling of the distribution of pesticides in ponds. NRCC Publ. No. NRCC 16073. Ottawa. p. 253-278.
- Mathieu, P. and M. Provancher. 1978. Étude des impacts des pulvérisations aériennes d'insecticide sur les écosystèmes aquatiques - le Lac Ste-Anne - Printemps 1977. Groupe conseil.
- Mathieu, P., P. Larochelle, and R. Dostie. 1979. Comportement des insecticides dans les milieux aquatiques, lors des pulvérisations aériennes contre la tordeuse des bourgeons de l'épinette. Région de St. Pascal de Kamouraska. Groupe Conseil Écologie, Dir. Domaine Hydrique, Min. Richesse Natur. 57 p.
- McLeese, D. W., V. Zitko, and D. B. Sergeant. 1979. Uptake and excretion of fenitrothion by clams, *Mya arenaria* and mussels, *Mytilus edulis*. Bull. Environ. Contam. Toxicol. 22: 800-806.
- Metcalf, C. D., D. W. McLeese, and V. Zitko. 1980. Rate of volatilization of fenitrothion from fresh water. Chemosphere. 9: 151-156.
- Mikami, N., K. Imanishi, H. Yamada, and J. Miyamoto. 1985. Photodegradation of fenitrothion in water and on soil surface and its hydrolysis in water. J. Pest. Sci. 10: 263-272.
- Miyamoto, J. 1977. Degradation of fenitrothion in terrestrial and aquatic environments including photolytic and microbial reactions. In NRCC Publ. No. NRCC 16073, Ottawa. p. 105-134.
- Miyamoto, J., N. Mikami, K. Mihara, Y. Takimoto, H. Kohda, and H. Suzuki. 1978. Biological activity of fenitrothion and its degradation products. J. Pestic. Sci. 3: 35-41.
- Miyashita, M. 1984. Effects of a short period exposure of fenitrothion on the reproduction of pregnant females of the guppy, *Poecilia reticulata*. Jap. J. of Hygiene. 39 (3): 647-650.
- Monenco Limited. 1982. Aminocarb and fenitrothion in aquatic plants and benthic macro invertebrates. Report for Environment New Brunswick, Monenco Limited, Fredericton, New Brunswick. 16 p.

- Montreal Engineering Company, Limited. 1976. Assessment of effects of selected insecticide application on drift of stream macro invertebrates. Report for New Brunswick Department of Natural Resources, Montreal Engineering Company, Limited, Fredericton, New Brunswick. 18 p.
- Montreal Engineering Company, Limited. 1978. Assessment of the effects of the 1977 New Brunswick spruce budworm control program on fish food organisms and fish growth. Report for New Brunswick Department of Natural Resources, Montreal Engineering Company, Limited, Fredericton, New Brunswick. 56p.
- Montreal Engineering Company, Limited. 1980. Effects on fish of the 1979 New Brunswick spruce budworm control program - aerial spraying. Montreal Engineering Company, Limited, Fredericton, New Brunswick. 40 p.
- Moody, R. P., R. Greenhalgh, L. Lockhart, and P. Weinberger. 1978. The fate of fenitrothion in an aquatic ecosystem. Bull. Environ. Contam. Toxicol. 19: 8-14.
- Morin, R., G. Gaboury, G. Mamarbachi, and R. Dostie. 1983. Résidus de fenitrothion et d'aminocarbe dans l'environnement. Ministère de l'Énergie et des Ressources, Québec, 60 p.
- Morin, R., G. Gaboury, and G. Mamarbachi. 1986. Fenitrothion and aminocarb residues in water and balsam fir foliage following spruce budworm spraying programs in Québec, 1979 to 1982. Bull. Environ. Contam. Toxicol. 36: 622-628.
- Morrison, B. R. S. and D. E. Wells. 1981. The fate of fenitrothion in a stream environment and its effect on the fauna following aerial spraying of a Scottish forest. Sci. Total Environ. 19: 233-252.
- Muirhead-Thomson, R. C. 1973. Laboratory evaluation of pesticide impact on stream invertebrates. Freshw. Biol. 3: 479-498.
- Murty, A. S., B. R. Rajabhushanam, A. V. Ramani, and K. Christopher. 1983. Toxicity of fenitrothion to the fish *Mystus cavasius* and *Labeo rohita*. Environ. Pollut. Ser. A Ecol. Biol. 30: 225-232.
- National Research Council Canada. 1975. Fenitrothion: The effects of its use on environmental quality and its chemistry. NRCC Publ. No. NRCC 14104. Ottawa. 162 p.
- National Research Council Canada. 1977. Fenitrothion: The long-term effects of its use in the forest ecosystem. S.R. Roberts, R. Greenhalgh and W.K. Marshall (eds.). NRCC Publ. No. NRCC 16073. Ottawa. 628 p.
- National Research Council Canada. 1982. Aminocarb: The effects of its use on the forest and the human environment. NRCC. Pub. No. NRCC 18979. Ottawa.
- Needham, P. R. and R. L. Usinger. 1956. Variability in the macrofauna of a single riffle in Prosser Creek, California, as indicated by the Surber sampler. Hilgardia 24: 383-409.
- Ohmae, T., M. Uno, T. Okada, Y. Onji, I. Terada, and K. Tanigawa. 1981. Environmental behaviour of fenitrothion and its decomposition products after aerial application. J. Pestic. Sci. 6: 437-446.
- Pawar, K. R. and M. Katdare. 1983. Effect of Sumithion of the ovaries of fresh water fish *Garra mullya*. Curr. Sci. (Bangalore) 52: 784-785.
- Penney, G. H. 1971a. Effects of the 1971 forest spray program in Fundy National Park on aquatic insects in a small stream. Fish. Serv., Dept. Fish. For. Can. MS Rep. No. 71-20.
- Penney, G. H. 1971b. Report on the effects of the 1970 New Brunswick forest spraying on juvenile salmon and their food organisms. Fish. Serv., Dept. Fish. For. Can. MS Rep. No. 71-13. 41 p.
- Penney, G. H. 1971c. Summary report on the effects of forest spraying in New Brunswick in 1971, on juvenile Atlantic salmon and aquatic insects. Fish. Serv., Dept. Env. Can. Rep. for the Interdept. Comm. For. Spray. Operations. 7 p.
- Penney, G. H. and J. R. MacDonald. 1966. Preliminary report on the effects of the 1966 forest spraying on New Brunswick salmon. Res. Dev. Branch, Dept. Fish. Can., Halifax. 21 p.

- Peterson, R. H. 1974. Influence of fenitrothion on swimming velocities of brook trout, *Salvelinus fontinalis*. J. Fish. Res. Board Can. 31: 1757-1762.
- Peterson, R. H. 1976. Temperature selection of juvenile Atlantic salmon, *Salmo salar*, as influenced by various toxic substances. J. Fish. Res. Board Can. 33: 1722-1730.
- Peterson, R. H. and V. Zitko. 1974. Variations in insect drift associated with operational and experimental contamination with fenitrothion in New Brunswick. Fish. Res. Board Can. Tech. Rep. 439. 37 p.
- Poirier, D. G. 1986. The toxicity of four insecticides used in forest pest management in Canada to selected aquatic invertebrates. M.Sc. thesis, Univ. of Guelph, Guelph, Ontario. 85 p.
- Poirier, D. G. and G. A. Surgeoner. 1986. Evaluation of the toxicity of spruce budworm insecticides to aquatic invertebrates. Can. Tech. Rep. Fish. Aquat. Sci. 1462. p. 99-120.
- Rabeni, C. and K. E. Gibbs. 1976a. The effects of Sumithion on non-target stream insects in Maine, 1975. Life Sci. Agr. Exp. Stn., Univ. Maine Orono. Misc. Rep. 176. 13 p.
- Rabeni, C. and K. E. Gibbs. 1976b. Effects of Dylox, Matacil and Sumithion on non-target stream insects in Maine, 1975, p. 64-74. In U.S. For. Serv. 1975 Cooperative pilot control project of Dylox, Matacil, and Sumithion for spruce budworm control in Maine, U.S. For. Serv., Upper Darby, Pennsylvania.
- Resh, V. H. 1979. Sampling variability and life history features: basic considerations in the design of aquatic insect studies. J. Fish. Res. Board Can. 36: 290-311.
- Riche, L. G. and L. Coady. 1969. Effects of aerial spraying with Sumithion on salmon brooks in western and central Newfoundland - 1969 with special reference to aquatic insects. Can. Dept. Fish. For., Res. Dev. Branch. Prog. Rep. 61. 43 p.
- Ripley, B. D., J. A. Hall, and A. S. Y. Chau. 1974. Determination of fenitrothion, phosphamidon, and dimethoate in natural water. Environ. Lett. 7: 97-118.
- St. Jean, R. and G. Mamarbachi. 1977. Résidus de fenitrothion, dimethoate, phosphamidon, et trichlorfon dans l'eau de certain ruisseaux du Québec et du fenitrothion dans les aiguilles de sapin baumier (arrosage, été 1975). Gouvernement du Québec, Service de protection de l'Environnement. 24 p.
- Sanders, H. O., M. T. Findley, and J. B. Hunn. 1983. Acute toxicity of 6 forest insecticides to 3 aquatic invertebrates and 4 fishes. U. S. Fish. Wildl. Serv. Tech. Pap. No. 110. p. 1-5.
- Sarrazin, R. 1976. Enquête sur la mortalité d'alevins de pisciculture dans la région de La Tuque, Québec, en 1975. Ministère du Tourisme, de la Chasse et de la Pêche, Service de la Recherche Biologique, Québec. 9 p.
- Sarrazin, R. 1977. Effets environnementaux des pulvérisations aériennes contre la tordeuse des bourgeons de l'épinette au Québec - 1977. Direction de la recherche faunique. Rapport de Recherche Faunique No. 15. 31 p.
- Saxena, P. K. and M. Garg. 1978. Effect of insecticidal pollution on ovarian recrudescence in the fresh water teleost *Channa punctatus*. Indian J. Exp. Bio. 16: 689-691.
- Saxena, P. K. and K. Mani. 1985. Protein-bound iodine levels in the blood plasma of freshwater teleost, *Channa punctatus* (Bl.) Exposed to subtoxic pesticide concentrations. Toxicol. Lett. 24: 33-36.
- Saxena, P. K., K. Mani, and J. K. Kondal. 1986. Effect on safe application rate concentrations of some biocides on the fresh-water murrel, *Channa punctatus*, a biochemical study. Ecotoxicol. Environ. Saf. 12: 1-14.
- Scherer, E. 1975. Avoidance of fenitrothion by goldfish, *Carassius auratus*. Bull. Environ. Contam. Toxicol. 13: 492-496.
- Scherer, E. and R. E. McNicol. 1986. Behavioral responses of stream-dwelling *Acroneturia lycorias* (Insecta: Plecoptera) larvae to methoxychlor and fenitrothion. Aquat. Toxicol. 8: 251-263.

- Sergeant, D. B. and V. Zitko. 1979. The determination of trichlorfon, dichlorvos, fenitrothion, and phosphamidon in water. Fish. Mar. Serv. Tech. Rep. No. 886. p. 1-12.
- Sergeant, D. B., V. Zitko, and L. E. Burrige. 1979. The determination of fenitrothion in bivalves. Fish Mar. Serv. Tech. Rep. No. 906. 27 p.
- Sexsmith, W. A. 1987. The evolution of buffer zones for forest insect spraying in New Brunswick p. 4-12 *In* Buffer Zones: Their application to forest insect control operations. Proc. Buffer Zone Workshop, Eastern Spruce Budworm Council's Environmental Committee, Québec City 16-17 April, 1986. 124pp. ISBN 0-662-15732-X.
- Sharma, R. K., S. Sharma, and S. Shandilya. 1981. Effect of Sumithion on the mortality rate of *Clarias batrachus*. Geobios(Jodhpur). 8:237-239.
- Sundaram, K. M. S. 1973. Degradation dynamics of fenitrothion in aqueous systems. Can. For. Serv., Chem. Control Res. Inst. Inf. Rep. CC-X-44. 19 p.
- Sundaram, K. M. S. 1974a. Persistence studies of insecticides: III. accumulation of fenitrothion and its oxygen analog in foliage, soil, and water in Larose Forest. Can. For. Serv., Chem. Control Res. Inst. Inf. Rep. CC-X-65. 21 p.
- Sundaram, K. M. S. 1974b. Distribution and persistence of fenitrothion residues in foliage soil and water in Larose Forest. Can. For. Serv., Chem. Control Res. Inst. Inf. Rep. CC-X-64. 43 p.
- Sundaram, K. M. S., P. D. Kingsbury, and S. B. Holmes. 1984. Fate of chemical insecticides in aquatic environments. p.253-276. *In* W. Y. Garner and J. Harvey jr. (ed.). Chemical and biological controls in forestry. A.C.S. Symp. Ser. 238. 397 p.
- Sur, R. K., K. C. Ghose, and A. Sur. 1979. Effects of some insecticides on the development of the catfish *Heteropneustes fossilis*. Indian Biol. 11: 53-56.
- Symons, P. E. K. 1973. Behaviour of young Atlantic salmon (*Salmo salar*) exposed to or force-fed fenitrothion, an organophosphate insecticide. J. Fish. Res. Board Can. 30: 651-655.
- Symons, P. E. K. 1977a. Dispersal and toxicology of the insecticide fenitrothion. Residue Rev. 38: 1-36.
- Symons, P. E. K. 1977b. Assessing and predicting effects of the fenitrothion spray program on aquatic fauna. NRCC Publ. No. NRCC 16073. p. 391-413.
- Symons, P. E. K. and G. D. Harding. 1974. Biomass changes of stream fishes after forest spraying with the insecticide fenitrothion. Fish. Res. Board Can. Tech. Rep. 432. 47 p.
- Symons, P. E. K. and J. L. Metcalfe. 1978. Mortality, recovery, and survival of larval *Brachycentrus numerosus* (Trichoptera) after exposure to the insecticide fenitrothion. Can. J. Zool. 56: 1284-1290.
- Takimoto, Y. and J. Miyamoto. 1976. Studies on the accumulation and metabolism of Sumithion in fish. J. Pestic. Sci. 1: 261-272.
- Takimoto, Y., S. Hugino, H. Yamada, and J. Miyamoto. 1984a. The acute toxicity of fenitrothion to killifish (*Oryzias latipes*) at twelve different stages of its life history. J. Pestic. Sci. 9: 463-470.
- Takimoto, Y., M. Ohshima, M. Yamada, and J. Miyamoto. 1984b. Fate of fenitrothion in several developmental stages of the killifish (*Oryzias latipes*). Arch. Environ. Contam. Toxicol. 13: 579-587.
- Takimoto, Y., M. Ohshima, and J. Miyamoto. 1987. Comparative metabolism of fenitrothion in aquatic organisms I. Metabolism in the euryhaline fish, *Oryzias latipes* and *Mugil cephalus*. Ecotoxicol. Environ. Saf. 13: 104-117.
- Thellan, C. 1982. Effets toxiques d'une preparation operationelle de fenitrothion sur trois espèces de salmonides. Éco-recherches Inc., Pointe Claire, Québec. 78 p.

- Thellan, C., R. Van Coillie, and N. St. Louis. 1982. Evaluation toxicologique d'une preparation operationnelle de fenitrothion sur des salmonides. Éco-recherches Inc., Pointe Claire, Québec. 26 p.
- Thirumurthi, S., P. Palaniswamy, R. R. Subramanian, and S. Kannammal. 1975. Effect of aerial spraying on non-target organisms inhabiting rice swamps. *Madras Agric. J.* 60: 573-574.
- Tracy, H. B. and B. Purvis. 1976. Aquatic monitoring of the 1975 spruce budworm fenitrothion aerial spray project in Washington State. Dept. of Ecology, Olympia, Washington, DOE-76-2. 34 p.
- Varty, I. W. (ed.). 1980. Environmental surveillance in New Brunswick, 1978-79. Effects of spray operations for forest protection against spruce budworm. Committee for Environmental Monitoring of Forest Insect Control Operations (EMOFICO), University of New Brunswick, Fredericton, N. B. 76p.
- Vincent, F. 1969. Observation of the effects of Sumithion on aquatic forms, spruce budworm control project, Maine. In: Cooperative Sumithion pilot study on spruce budworm - 1968. USDA Forest Service, USDI, State of Maine. p. 28-37.
- Volpé, G. and V. N. Mallet. 1979. Recovery of fenitrothion from water using Amberlite resin XAD-7. *J. Chromatogr.* 177: 385-389.
- Volpé, G. and V. N. Mallet. 1980. Development of an analytical method for fenitrothion and five derivatives in water using XAD resins and gas liquid chromatography (GLC). *Int. J. Env. Anal. Chem.* 8: 291-301.
- Volpé, G. and V. N. Mallet. 1981. High-performance liquid chromatography of fenitrothion and seven derivatives - a study of their recovery from water using XAD resins as compared with organic solvents. *Chromatogr.* 14: 333-336.
- Weinberger, P., R. Greenhalgh, R. P. Moody, and B. Boulton. 1982a. Fate of fenitrothion in aquatic microcosms and the role of aquatic plants. *Environ. Sci. Technol.* 16: 470-473.
- Weinberger, P., R. Greenhalgh, D. Sher, and M. Ouellette. 1982b. Persistence of formulated fenitrothion in distilled estuarine and lake water microcosms in dynamic and static systems. *Bull. Environ. Contam. Toxicol.* 28: 484-489.
- Weinberger, P., D. Sher, and R. Greenhalgh. 1983. Interaction between co-solvents and algae in the residue dynamics of fenitrothion. *J. Environ. Sci. Health Part B Pestic. Food Contam. Agric. Wastes.* 18: 269-282.
- Weiss, C. M. 1961. The determination of cholinesterase in the brain tissue of three species of fresh water fish and its inactivation *in vivo*. *Ecology* 39: 194-199.
- Wells, D. E., B. R. S. Morrison, and A. A. Cowan. 1978a. Chemical and biological observations in streams following aerial spraying of forests with fenitrothion, p. 103-131. In A. V. Holden and D. Bevan [eds.]. Control of the pine beauty moth by fenitrothion in Scotland. Forestry Commission, Nuffield Press, Oxford.
- Wells, P. G., R. A. Matheson, and D. A. Lord. 1978b. Monitoring of the 1978 New Brunswick spray program, field sampling and aquatic toxicity studies. *For. Pest. Con. Forum., Can. For. Serv., Ottawa*, 1978. 12 p.
- Wells, P. G., R. A. Matheson, D. A. Lord, and K. G. Doe. 1979. Environmental monitoring of the 1978 spruce budworm spray program in New Brunswick, Canada - field sampling and aquatic toxicity studies with fish. *Env. Can. Surv. Rep. EPS-5-AR-79-1.* 14 p.
- Wildish, D. J. 1974. Total body lipid and brain acetylcholinesterase activity of brook trout exposed to fenitrothion. *Fish. Res. Board Can. MS Rep.* 1306.
- Wildish, D. J., W. G. Carson, T. Cunningham, and N. J. Lister. 1971. Toxicological effects of some organophosphate insecticides to Atlantic salmon. *Fish. Res. Board Can. MS Rep.* 1157. 22 p.
- Wildish, D. J. and R. L. Phillips. 1972. Acute lethality of fenitrothion to freshwater aquatic insects. *Fish. Res. Board Can., MS Rep.* 1210. 11 p.
- Wildish, D. J. and N. A. Lister. 1973. Biological

- effects of fenitrothion in the diet of brook trout. Bull. Environ. Contam. Toxicol. 10: 333-339.
- Wildish, D. J. and N. A. Lister. 1979. Effects of dietary fenitrothion on growth and hierarchical position in brook trout. Prog. Fish-Cult. 39: 3-9.
- Wiley, M. and S. L. Kohler. 1984. Behavioral adaptations of aquatic insects. p. 101-133. In V. H. Resh and D. M. Rosenberg (eds.). The ecology of aquatic insects. Praeger Publishers, New York. 625 p.
- Witty, M. J. 1985. Cholinesterase activities in brook trout: a bioassay study of fenitrothion containing Triton X-114 on brook trout. Unpublished, Sault College of Applied Arts and Technology. 24 p.
- Woodward, D. F. and W. L. Mauck. 1980. Toxicity of five forest insecticides to cutthroat trout and two species of aquatic invertebrates. Bull. Environ. Contam. Toxicol. 25: 846-853.
- Yasuno, N., S. Hatakeyama, and M. Miyashita. 1980. Effects of reproduction in the guppy, *Poecilia reticulata*, under chronic exposure to fenitrothion. Bull. Environ. Contam. Toxicol. 25: 29-33.
- Yasuno, M., J. Okita, K. Saito, Y. Nakamura, S. Hatakeyama, and S. Kasuga. 1981. Effects of fenitrothion on benthic fauna in small streams of Mount Tskuba, Japan. Jap. J. Ecol. 31: 237-246.
- Yunus, A. and L. G. Soon. 1971. A problem in the use of insecticides in paddy fields in West Malaysia, a case study. Malays. Agric. J. 48: 167-183.
- Zakrevsky, J. G. and V. N. Mallet. 1977. An *in situ* fluorometric method for the detection and quantitative analysis of fenitrothion, its breakdown products and amine-generating compounds. J. Chromatogr. 132-315.
- Zitko, V., W. V. Carson, and B. J. Findlayson. 1970. The inhibition of fish brain acetylcholinesterase activity by fenitrothion, Bay 77488, and Dylox, and by the 1969 aerial spraying of fenitrothion in New Brunswick. Fish. Res. Board Can. MS Rep. 1108.
- Zitko, V. and T. D. Cunningham. 1974. Fenitrothion, derivatives and isomers: hydrolysis, adsorption, and biodegradation. Fish. Res. Board Can. Tech. Rep. 458. 27 p.
- Zitko, V. and T. D. Cunningham. 1975. Fish Toxicity of S-methyl fenitrothion. Bull. Environ. Contam. Toxicol. 14: 19-24.